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Diagnostic Feasibility Study  
of Lake of the Woods  
Marshall Co., Indiana

Final Report  
to the  
U.S. Environmental Protection Agency  
February 28, 1982

Prepared and submitted by  
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and  
Kenneth E. Roberts

Department of Biology  
Ball State University

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Indiana State Board of Health, Division of Water Pollution Control

Indiana Department of Natural Resources, Tri-Lakes Fisheries Station

Lake of the Woods Lake Association

Ball State University, Office of Research and Contracts and Grants Office

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## ABSTRACT

Lake of the Woods is a glacial kettle lake in north central Indiana, with a surface area of 164 ha. and a volume of  $7.85 \times 10^6 \text{ m}^3$ . Limnological studies conducted from September, 1980, through August, 1981, allowed formulation of hydrologic and nutrient budgets. Mass water loading to the lake was  $6.69 \times 10^6 \text{ m}^3 \text{ y}^{-1}$ , and resulted in a hydraulic residence time,  $t_w$ , of 1.17 years. Approximately 71% of the water entered the system via streamflow input. Measured external areal phosphorus and nitrogen loadings were  $0.62$  and  $49.7 \text{ g m}^{-2} \text{ y}^{-1}$ , respectively. Streamflow input of nutrients contributed 59% of the external P and 92% of the external N loading. Septic input and internal loading of nutrients from the sediments was significant. Lake of the Woods retained 70% of the P and 52% of the N entering the system. The mean annual P concentration was  $65 \mu\text{g l}^{-1}$ , indicative of strong eutrophication. Management strategies focus on reducing internal loading, septic inputs, and runoff from agricultural lands.

## CONCLUSIONS

1. Lake of the Woods is a eutrophic lake with poor water quality.
2. Annually, approximately 71% of the water entering the Lake of the Woods system comes from streamflow input.
3. Major external sources of phosphorus were streamflow input (59.3% of external loading) and septic loading (22.2% of external loading).
4. The nitrogen:phosphorus ratio for external loading is 81:1.
5. The internal loading of phosphorus from the sediments was very significant, comprising 42% of the annual phosphorus loading to Lake of the Woods.
6. Plant communities of Lake of the Woods are phosphorus limited.
7. Macrophyte and algal communities in Lake of the Woods are excessive, and present serious water quality problems.
8. Bacteriological analysis of Lake of the Woods indicates that serious public health problems exist.
9. Septic systems are negatively impacting the lake, and should be corrected by the installation of a sewerage system.
10. Management strategies focus on cosmetic control of macrophytes (i.e. weed barriers, mechanical harvesting, and herbicides), reducing in-lake phosphorus concentration and sealing the sediments (Alum treatment), reducing external loading of phosphorus (no-till agriculture, grassed waterways), sewerage the lake community, and other long term strategies (wetlands and development regulation).

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## I. INTRODUCTION

Lake of the Woods was selected for a diagnostic feasibility study leading to a lake restoration-management effort because of its eutrophic state and concurrent problems for recreational uses. Furthermore, public support among lake residents for this project was strong, and the lake was given high priority for such a study by the Indiana State Board of Health. In fall, 1980, following approval and funding by the U.S. Environmental Protection Agency, the project was initiated by Ball State University. Intensive tributary and in-lake monitoring was conducted through August, 1981. This sampling allowed for the development of extensive nutrient budgets for the lake. Using the information in these budgets, a management plan for restoring the water quality of Lake of the Woods has been developed. This plan has been reviewed by lake residents and its initiation awaits funding by state and/or federal agencies.

## II. PHYSICAL CHARACTERISTICS

### A. Morphometric Measurements

Lake of the Woods is a medium size lake of 164.29 hectares (406 acres) located in Marshall County, Indiana (Fig. 1). Morphometric information for the lake was obtained from a 1957 USGS bathymetric map of the lake (Fig. 2). Using the methods of Wetzel and Likens (1979), morphometric parameters were calculated by planimetry and these are summarized in Table 1. Hypsographic and depth volume curves were also plotted, and these are shown in Figure 3.

Lake of the Woods has a maximum depth of 14.6 m and a mean depth of 4.78 m. The lake has a relative depth of 1.01% which means that the lake is about average in its resistance to wind mixing. The shoreline development,  $D_L$ , of 1.46 indicates that the lake is roughly oval in shape with few bays and/or coves along its shore.

### B. Tributaries

Lake of the Woods receives input from six streams. Five of these streams are located on the western shore of the lake while the sixth is located on the eastern shore (Figs. 2,4). All six of these tributaries represent drainage ditches that channel water into the lake from surrounding farm fields. The exact drainage area of each tributary is hard to determine due to the complex pattern of field tiles and runoff ditches that have been constructed by farmers to reduce standing water on their fields. It can be safely stated, though, that all six ditches are heavily impacted by agricultural runoff.

Lake of the Woods is drained by a single outflow at the southern tip of the lake (Figs. 2,4). This outflow is regulated by a dam which can be adjusted to allow the lake to drain during periods of high water. The flow in the outflow stream has varied considerably throughout the study period.

#### C. Lake Use

The size of the Lake of the Woods watershed is  $2.4501 \times 10^7 \text{ m}^2$ . The dominant land usage of the watershed is cultivated land (75%) and the major crop is corn. About 10% of the land is forested, while 9% remains in hay or pasture usage. Miscellaneous land uses account for about 6% of the watershed area. This leaves less than 1% of the land as wetland and commercial services (marina, camps, etc.).

The agricultural practices of the farmers in the watershed are similar to most farming techniques used in the mid-west. Fields are often plowed in late fall after harvest and allowed to remain barren all winter. Heavy fertilizer applications of phosphorus and ammonia are made prior to spring planting. Herbicides are applied at or soon after planting.

Fig. 1. Location of Lake of the Woods.

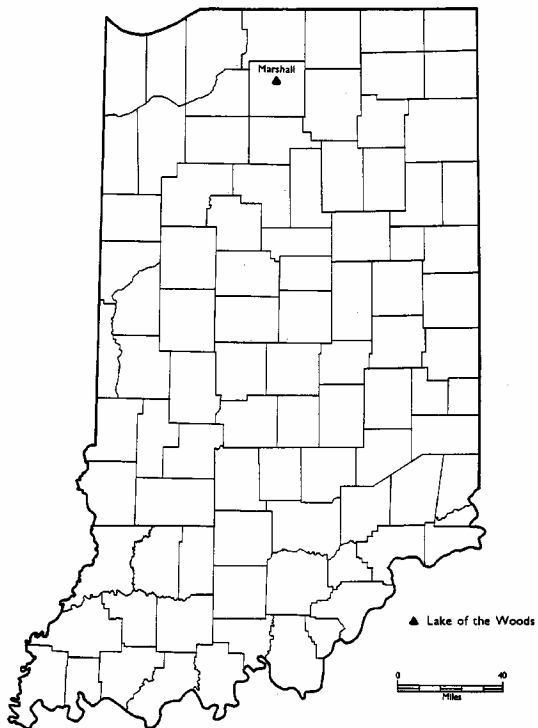


Fig. 2. Bathymetric map of Lake of the Woods (after U.S.G.S., 1957).  
Contours are in 5 ft intervals.

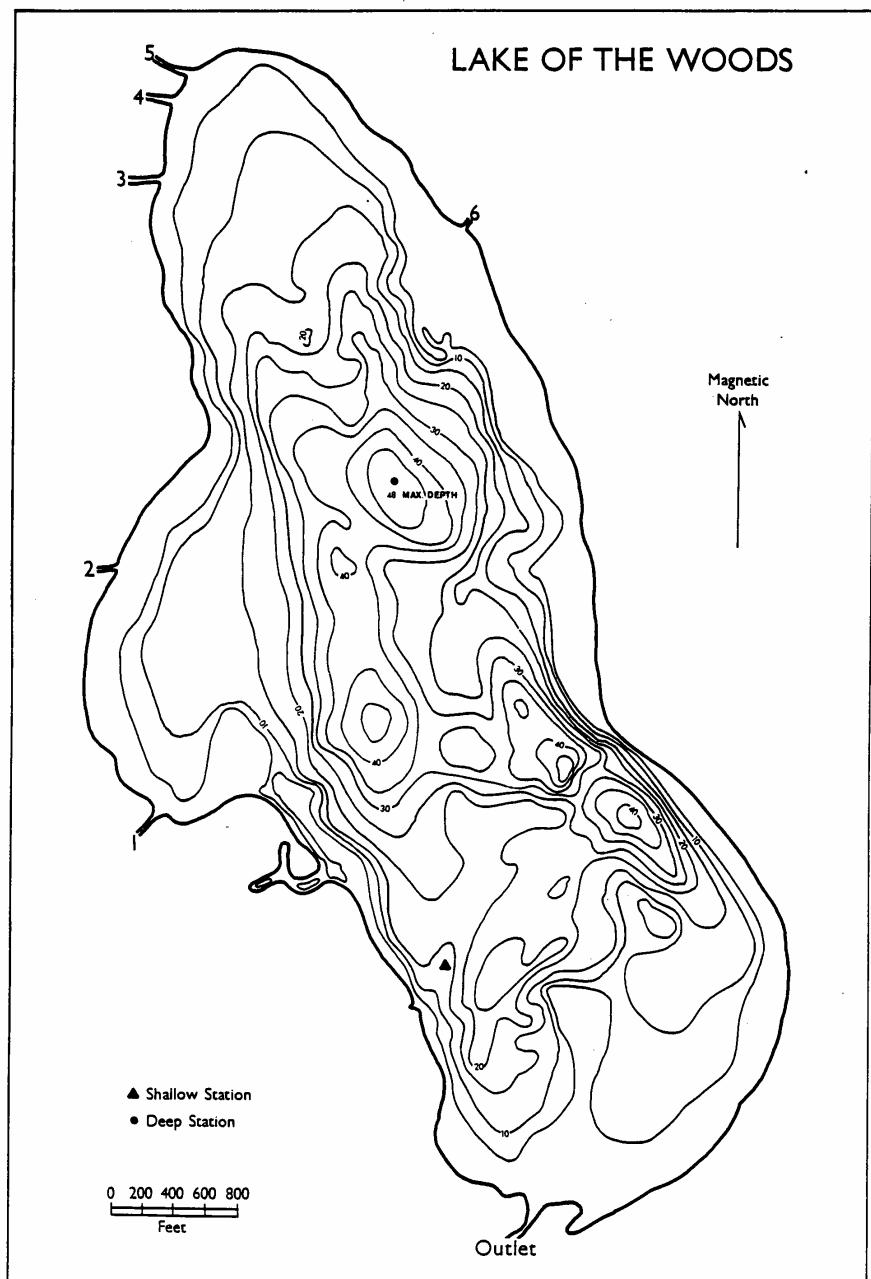


Fig. 3. Hypsographic (A) and depth-volume (B)  
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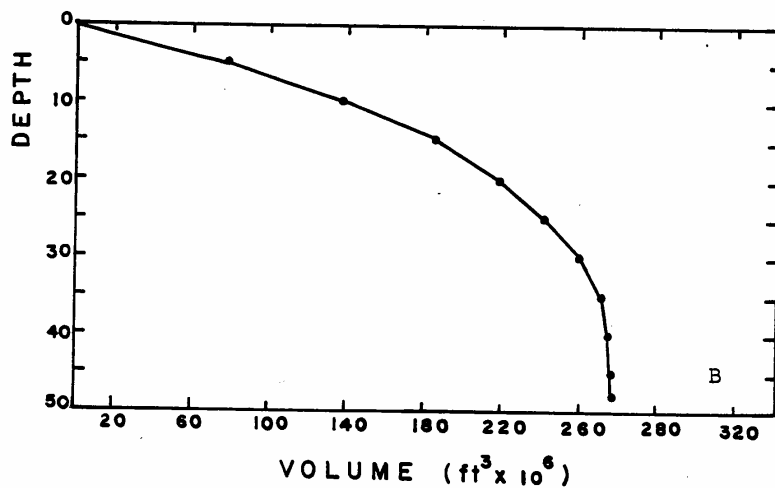
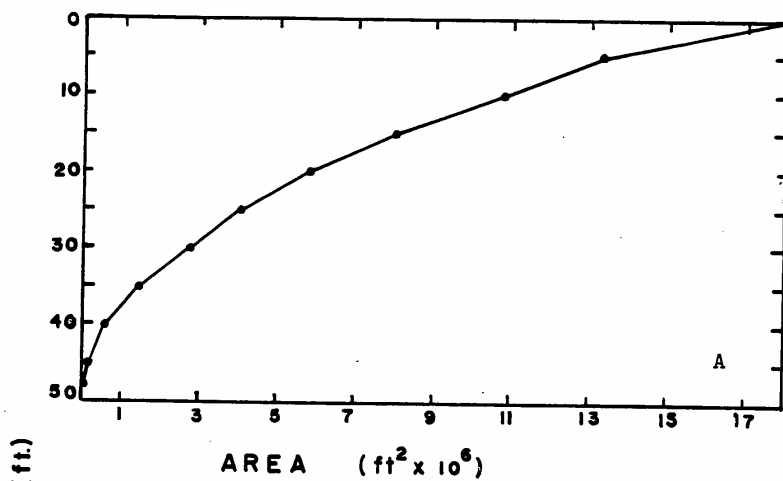


Fig. 4. Diagrammatic view of Lake of the Woods watershed, showing tributary sampling stations.

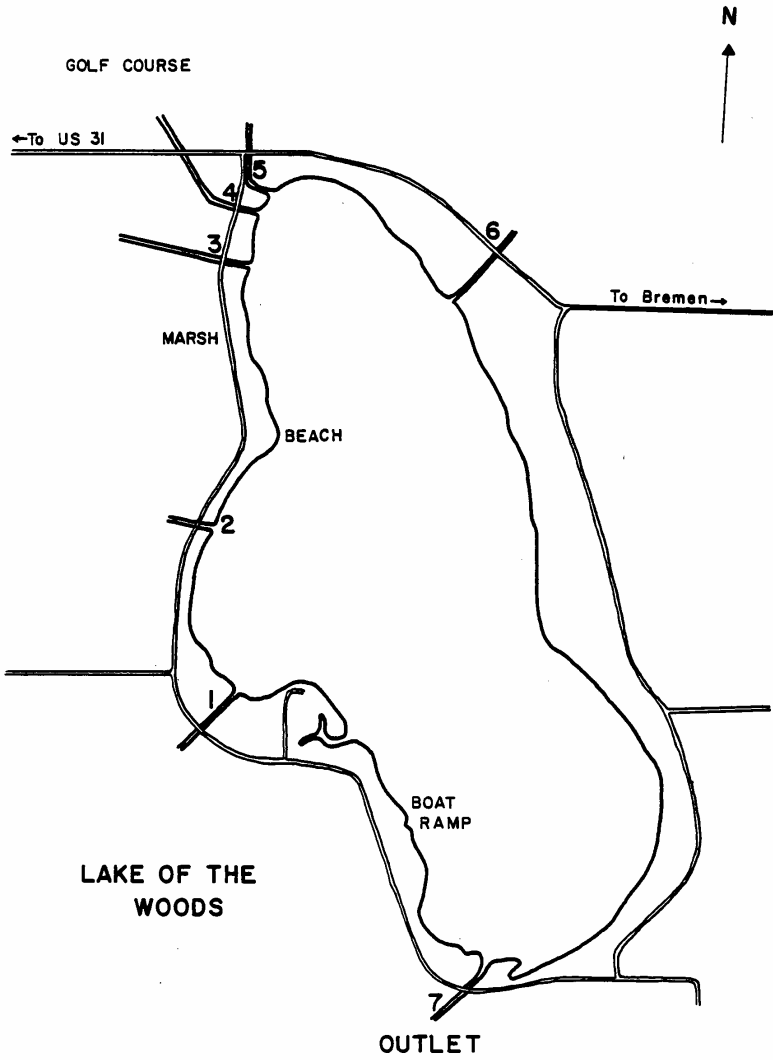


TABLE 1.

Morphometric Parameters for Lake of the Woods.

Parameter	Symbol	Value
Surface area	$A_o$	164.29 ha
Volume	$V$	7,848,057 m <sup>3</sup>
Shoreline length	$L$	6631.8 m
Shoreline development index	$D_L$	1.46
Volume development index	$D_V$	0.98
Maximum effective length	$l$	2408.3 m
Maximum breadth	$b$	1032.1 m
Mean breadth	$\bar{b}$	682.2 m
Maximum depth	$Z_m$	14.6 m
Mean depth	$\bar{Z}$	4.78 m
Relative depth	$Z_r$	1.01 %
Center of gravity	$Z_g$	3.17 m

### III. METHODS

#### A. Tributary and Outlet monitoring

The six inlets and single outlet of Lake of the Woods were monitored from November, 1980, through August, 1981. The outlet stream was sampled at a location about 20 m downstream from the lake (Site 7). The first tributary was sampled about 15 m from its entrance to the lake (Site 1). The second tributary was sampled about 3 m from its entrance to the lake, adjacent to a residential property (Site 2). Tributary #3 was sampled on the western side of the culvert which runs beneath West Shoreline Drive; this location, Site 3, was situated about 8 m upstream from a large wetland area through which the stream passes as it enters the lake. The fourth tributary was sampled 5 m upstream (west) of the culvert which runs beneath West Shoreline Drive (Site 4). Sampling Site 5 was located in the fifth tributary ditch adjacent to a trailer park on the east shore of the stream and approximately 100 m upstream from the lake. The sixth tributary was sampled on the northern side of a culvert which runs beneath North Shoreline Drive and connects the ditch directly to the lake (Site 6). The position of each sampling site is shown in Figure 4.

##### 1. Chemical variables

Water samples for chemical analyses were collected at all tributary sampling locations (Sites 1-7) during the sampling period. Water samples were analyzed for total P, soluble-reactive P,  $\text{NH}_3$ ,  $\text{NO}_2$ ,  $\text{NO}_3$ , organic N, silica, alkalinity, total and particulate organic matter, and total and organic residue. Samples were collected in 1-2 Nalgene bottles, preserved with two ml of 10.8 N  $\text{H}_2\text{SO}_4$  (for appropriate parameters), placed in an ice chest, and returned to the laboratory for

analysis. Measurements of soluble reactive P were made immediately upon return to the laboratory; all other nutrient assays were completed within 24-48 hours after sample collection. Sequential samplers (Isco Model 1680) were placed at Sites 4, 5, and 7 to obtain daily integrated water samples (a sample was collected every 6 hours); these were collected at biweekly intervals through June and July and returned to the laboratory. Analytical procedures for determinations of the chemical variables outlined above were identical to the procedures used in the lake monitoring of chemical variables. These procedures are outlined in detail in Appendix I.

## 2. Streamflow

The volume of water carried in each tributary stream and the outlet was determined from measurements of water height. At all sampling sites lacquered meter sticks were mounted in the streams to serve as staff gauges, and the height of the stream was recorded several times weekly by local residents. Beginning in April, 1981, sampling Sites 4 and 5 were monitored continuously using Leopold Stevens Type F water level recorders. The water level recorders were enclosed in weather- and vandal-proof boxes and mounted in stilling wells consisting of culvert pipe 30.5 cm in diameter. Continuous water level readings were recorded on eight-day charts and calibrated to water level using the lacquered meter stick staff gauges.

On several dates throughout the year, corresponding to various water level heights, stream flow measurements were taken at all sampling sites using a Marsh-McBirney Model 201 Flow Meter. A smooth-bottomed area of the stream was selected and flow measurements

were made at 60% of the water depth for each 0.25 m width interval across the stream. Total stream flow (as  $\text{m}^3 \text{s}^{-1}$ ) was calculated according to the mid-section method of Nemerow (1974) using a computer program written by H. Senft (Appendix I).

## B. Lake monitoring

In-lake monitoring of Lake of the Woods commenced in late November, 1980, and continued through August, 1981. The lake was sampled monthly from November to March. However, unsafe ice conditions on the lake in December and February prevented sampling during those months. Beginning in early April and continuing through August, the lake was sampled biweekly. Several physical, chemical and biological variables were measured on each sampling date. Water samples for chemical analyses were collected in 1 l Nalgene bottles, preserved with two ml of 10.8 N  $\text{H}_2\text{SO}_4$  (appropriate parameters only), placed in an ice chest, and returned to the laboratory for analysis. Selected water samples for each chemical variable were triplicated to obtain estimates of precision. Methods of analysis for each of these variables are discussed individually below.

### 1. Physical variables

#### Temperature

In situ temperature profiles were obtained at 0.5 m intervals at both the deep and shallow stations using a Hydrolab 4041 probe. On occasion, a YSI telethermometer was used in place of the Hydrolab 4041 probe.

#### Conductivity

In situ conductivity measurements were made using a Hydrolab 4041 probe. Readings were taken at 0.5 m intervals from the surface to the bottom at both in-lake stations.

### Turbidity

Depth profiles of turbidity were obtained at both the deep and shallow water stations of Lake of the Woods. Measurements were made using a H. F. Instruments Model DRT-15 Series A turbidameter on water samples which were returned to the laboratory, warmed, and shaken.

### Secchi Disk

On each sampling date, Secchi disk readings were obtained at both the shallow and deep water lake stations. A 20 cm diameter standard Secchi disk was lowered into the lake until it became invisible and then raised slowly until it reappeared; this depth was noted and recorded as the Secchi disk depth.

### Light Attenuation

The attenuation of photosynthetically active radiation (Ph.A.R.) in Lake of the Woods was measured using a LICOR Model LI-186A quantum sensor fitted with a  $4\pi$  spherical collector. Depth profiles of light intensity were taken by lowering the quantum sensor through the water column and obtaining readings at discrete 0.25 or 0.5 m intervals. Attenuation coefficients were calculated according to Lambert-Beers relationship

$$I_{z_j} = I_{z_i} e^{-nz_{ij}} \quad 1)$$

where

$$I_{z_j} = \text{intensity at depth } z_j \text{ } (\mu\text{Ein m}^{-2}\text{s}^{-1})$$

$$I_{z_i} = \text{intensity at depth } z_i \text{ } (\mu\text{Ein m}^{-2}\text{s}^{-1})$$

$$n = \text{total attenuation coefficient } (\text{m}^{-1})$$

$$z_{ij} = \text{depth interval } z_i \text{ to } z_j \text{ (m)}$$

Total attenuation coefficients,  $n$ , were partitioned into two components: a portion due to chlorophyll a ( $n_c$ ) and a portion due to all other factors ( $n_w$ ). Equation 1 was thus re-written as:

$$I_{z_j} = I_{z_1} e^{-(n_c C + n_w) z_{1,j}} \quad 2)$$

where  $C$  is the chlorophyll a concentration of the water ( $\text{mg m}^{-3}$ ) and  $n_c$  is a constant  $\{0.016 \text{ m}^3 (\text{mg chl } \underline{a})^{-1}\}$ .

Replicate or triplicate profiles were averaged to calculate attenuation coefficients. All measurements were taken under constant light conditions.

## 2. Chemical variables

### pH

In situ profiles of pH were obtained at 0.5 m depth intervals at both the deep and shallow stations using a Hydrolab 4041 probe. The probe was calibrated before and after each use according to manufacturer specifications.

### Dissolved oxygen

In situ profiles of dissolved oxygen were obtained at 0.5 m depth intervals at both the deep and shallow stations using a Hydrolab 4041 probe. This probe was calibrated before and after each use according to manufacturer specifications.

### Phosphorus

#### Total

Laboratory determinations of total phosphorus were made on acidified water samples. A 50 ml aliquot was analyzed using the persulfate digestion procedure of Menzel and Corwin (1965) followed

by the molybdate-blue spectrophotometric determination with ascorbic acid as the reductant. Complete details are presented in Procedure 2b of Appendix I.

#### Soluble Reactive

Soluble reactive phosphorus measurements were made immediately upon return to the laboratory. Water samples were filtered through pre-rinsed 0.45  $\mu$  glass filters (Gelman Type A/E). A 50 ml aliquot was analyzed spectrophotometrically using the ascorbic acid, molybdate-blue method. Details are outlined in Procedure 2c of Appendix I.

#### Nitrogen

##### Ammonia

Ammonia concentrations were determined in 50 ml aliquots of the acidified water sample using the method of Chaney and Marboch (1962). Following pH adjustment and reagent addition, spectrophotometric analysis of the sample was completed. Details of the method are given in Procedure 3 of Appendix I.

##### Nitrite

Nitrite was spectrophotometrically determined by the sulfanilamide diazotization technique of the E.P.A. (1979). A 25 ml subsample of acidified lake water was buffered and received the color reagent which reacted with nitrite to form an azo dye complex. Procedure 4 of Appendix I contains specific details of this method.

##### Nitrate

Spectrophotometric determination of nitrate was made by sulfanilamide diazotization following manual cadmium reduction of all

nitrate to nitrite (E.P.A. 1979). Background nitrite levels were subtracted from this reading to give nitrate values. Consult Procedure 5 of Appendix I for this methodology.

#### Organic

Water samples collected for organic nitrogen analysis were acidified and cooled. Measurements of organic nitrogen were made using the persulfate digestion technique of Raveh and Avnimelech (1979). All nitrogen was oxidized by persulfate digestion, subsequently reduced with DeVarda's Alloy, and spectrophotometrically determined as ammonia using Solorzano's (1969) technique. Ammonia, nitrite, and nitrate background levels were subtracted to yield the organic nitrogen fraction. Specific methodology is listed in Procedure 6 of Appendix I.

#### Reactive Silica

Silica determinations were made on non-acidified water samples using a modified molybdosilicate method (A.P.H.A. 1971). Water samples were frozen at  $-20^{\circ}\text{C}$  until analysis. After filtration with 0.45  $\mu$  Millipore HA filters and reagent addition, molybdosilicic acid concentrations were determined spectrophotometrically in 50 ml aliquots of water. Details of this technique are presented in Procedure 7 of Appendix I.

#### Residue

##### Total

Total residue was determined gravimetrically on non-acidified water samples using the technique recommended by the E.P.A. (1979). A known volume of water was evaporated in a drying oven and the total residual weight was obtained. Procedure 8 of Appendix I outlines the method.

### Volatile

The dessicated sample used for total residue analysis was ashed at  $550^{\circ}$  C for one hour. Weight loss upon combustion was determined and volatile residue was calculated (E.P.A. 1979). Refer to Procedure 8 of Appendix I for details.

### Particulate Matter

#### Total

Total particulate matter was measured by filtering a known water volume (500-1000 ml) through a preweighed  $0.45 \mu$  glass fiber filter (Gelman Type A/E). Filters were dessicated in a drying oven at  $101^{\circ}$  C and reweighed. Total particulate matter was calculated by difference (Appendix I; Procedure 9).

#### Organic

Particulate organic matter was determined by measuring weight loss of the dried filter upon combustion at  $550^{\circ}$  C for one hour in a muffle furnace (Appendix I; Procedure 9).

### Alkalinity

Total alkalinity was measured potentiometrically (Wetzel and Likens 1979). A 50 ml aliquot of non-acidified sample was titrated with 0.02 N sulfuric acid to a pH endpoint of 4.8. Total alkalinity was calculated from the volume of titrant used. A complete description of the method is contained in Procedure 10 of Appendix I.

## 2. Biological Variables

### Bacteria

Water samples for bacteriological analyses were collected monthly from April through August at lake and tributary stations. All

samples were collected aseptically from surface water in a 250 ml sterile screw cap bottle. Samples were immediately cooled on ice for return to the laboratory for processing. In the laboratory, water samples were diluted according to Standard Methods for the Examination of Water and Wastewater (A.P.H.A. 1975) using 90 ml buffered water blanks. Appropriate dilutions were selected for lake and tributary water according to recommendations given in Microbiological Methods for Monitoring the Environment (E.P.A. 1978). Membrane filtration techniques were employed for determining total coliform, fecal coliform, and fecal streptococci in the water samples. The procedure followed was that outlined in Microbiological Methods for Monitoring the Environment (E.P.A. 1978). Samples were passed through sterile filters of 0.45  $\mu$ m pore diameter (Gelman), placed in sterile petri dishes containing approximately 2 ml broth with a sterile absorbent pad of 3-5 ml agar. Duplicate samples were run of many dilutions.

Total coliform counts were determined by incubating membrane filters on sterile M-Endo broth (Difco) containing an absorbent pad. Plates were inverted, placed in a 35° C incubator and read after 24 hours. Those colonies showing a green metallic sheen under 10X magnification were considered positive. Those plates showing 20-80 colonies were chosen for determination of total coliform counts. Total coliforms were reported as numbers per 100 ml.

Fecal coliform counts were determined by incubating membrane filters on FC broth (Difco) for 24 hours at 44° C. Those plates

showing 20-60 colonies were examined under 10X magnification to determine the fecal coliform count. Fecal coliforms were reported as numbers per 100 ml sample.

Fecal streptococci counts were determined using KF agar (Difco) and incubation of 48 hours, at 35° C. Filters showing between 20-100 pink to dark red colonies were chosen and counted under 10X magnification. Fecal streptococci were reported as numbers per 100 ml sample.

Verification of total coliform, fecal coliform, and fecal streptococci counts were performed on 10% of the total samples. Methods for verification of these counts are presented in Procedures 11a, 11b, and 11c of Appendix I.

The remaining bacteriological analysis conducted was that of the standard plate count. Appropriate dilutions of water samples were placed on Difco plate count agar (A.P.H.A. 1975) and distributed by the spread plate technique. Plates were incubated at 35° C for up to 7 days. Duplicate plates were run of the various dilutions. After 2 and 7 day incubation, plates were removed and those showing between 30-300 colonies were counted. Counts were reported as cells per ml sample.

#### Chlorophyll a

Chlorophyll a concentrations of lake water were determined spectrophotometrically from the SCOR/UNESCO equations of Strickland and Parsons (1965). Samples were filtered onto 0.45  $\mu$  glass-fiber filters, ground in 90% acetone, and extracted for 10 minutes in the dark. More complete details are presented in Procedure 12 of Appendix I.

### Phytoplankton Communities

Depth profiles of algal samples were collected with a 6 l Van Dorn bottle at biweekly intervals at both the deep and shallow stations. Approximately 900 ml of lake water from each 1.5 m depth interval were preserved with 100 ml of F.A.A. solution in 1 l sample bottles, stored on ice, and returned to the laboratory. Each 1 l sample bottle was thoroughly shaken and exactly 250 ml was transferred to a 250 ml graduated cylinder. The remaining sample was measured and the total volume recorded. The 250 ml graduated cylinders were sealed with parafilm, labeled, and allowed to settle for four days. The samples were then concentrated to 50 ml by decanting the supernatant with a vacuum of 7 lbs Hg. The 50 ml concentrated samples were thoroughly mixed and transferred to labeled vials and kept refrigerated until counted.

Subsamples of the 50 ml concentrate were settled for 24 hours in settling chambers and counted at 100X to 320X on a Leitz Diavert Microscope. A minimum of 20-30 Whipple disc fields were counted to insure a large enough sample size. Counts were converted to numbers per ml using the formula:

$$\frac{\# \text{cells}}{\# \text{WD}} \times \frac{\# \text{WD}}{\text{SC}} \times \frac{\text{SC}}{\# \text{nl}} \times \frac{50 \text{ Ml}}{250 \text{ ml}} \times 1.11 = \text{cells ml}^{-1}$$

where

#cells = total cells counted

#WD = total Whipple disc areas counted

SC = settling chamber area

#ml = volume of subsample settled

1.11 = correction factor for preservative dilution

Algae were grouped into phylogenetic classes for determinations of percentage composition.

Cell volume estimations were made concurrently with cell counts by measuring algal cell dimensions, substituting these values into published formulae (Willen 1976), and computing average cell volumes. Calculated volumes were compared to published values for volume (Wetzel 1975) to check for agreement. Total cell volumes were calculated by multiplying the average cell volume of each species by the number of individuals per ml.

Algae were again grouped into phylogenetic classes for determinations of composition by volume.

#### Primary Production

Primary productivity measurements using the light-dark bottle method of oxygen production were conducted three times on Lake of the Woods during the summer. Lake water was collected at various depths with an opaque 6 l Van Dorn bottle. At each depth, duplicate transparent and opaque 300 ml B.O.D. bottles were filled and suspended in the lake. A third set of duplicate bottles was also filled at each depth for determinations of initial oxygen concentrations; these bottles were 'fixed' immediately with Winkler Reagents. The suspended bottles were incubated for 3-5 hours during mid-day. Light extinction was measured three times throughout the experiment.

At the completion of the incubation period, the bottles were retrieved from the lake, fixed with Winkler Reagents, and stored in light-tight boxes for return to the laboratory. Once at the

laboratory, the samples were acidified with concentrated  $\text{H}_2\text{SO}_4$ , and titrated with 0.025 N sodium thiosulfate using starch as an indicator according to the Winkler technique (A.P.H.A. 1975). The measured oxygen content of each bottle ( $\text{mg } \ell^{-1}$ ) was used to determine gross photosynthesis ( $\text{mg } \text{O}_2 \ell^{-1}\text{h}^{-1}$ ) according to the following formula:

$$\frac{L - D}{t}$$

where

L = light bottle oxygen concentration ( $\text{mg } \ell^{-1}$ )

D = dark bottle oxygen concentration ( $\text{mg } \ell^{-1}$ )

t = time ( $\text{h}^{-1}$ )

These values were converted to carbon fixation, using the correction factors of 0.375 mg C per mg  $\text{O}_2$  and an assimilation coefficient of 1.2 mg  $\text{O}_2$  evolution per 1.0 mg C fixation.

#### Algal Assay

As an aid in determining the limiting nutrient to phytoplankton growth in Lake of the Woods, the Environmental Protection Agency's Selenastrum capricornutum Printz Algal Assay Bottle Test was performed on water samples from Lake of the Woods. Growth of the test alga, S. capricornutum, was evaluated under several nutrient conditions. Surface lake water was collected in plastic jugs and stored on ice until processing.

Upon return to the laboratory, the lake water was autoclaved and filtered through 0.45  $\mu\text{m}$  Millipore filters. Numerous 250 ml flasks were filled with 150 ml of this water and stored for 24 hours to allow for  $\text{CO}_2$  equilibration. Experimental treat-

ments were run in triplicate: lake water alone, lake water +  $0.05 \text{ mg } \ell^{-1}$  phosphorus, lake water +  $1.0 \text{ mg } \ell^{-1}$  nitrate, and lake water +  $0.05 \text{ mg } \ell^{-1}$  phosphorus +  $1.0 \text{ mg } \ell^{-1}$  nitrate. Flasks filled with synthetic media served as controls for the experiment. Algal inoculum was prepared by phosphorus starving a seven day old culture of S. capricornutum in phosphorus-free media for 24 hours. Enough of this culture was added to each experimental flask to yield a density of ca.  $1000 \text{ cells ml}^{-1}$ . The flasks were incubated for 14 days at a temperature of  $24^{\circ} \text{C} \pm 1^{\circ} \text{C}$  and an illumination of  $500 \mu\text{Ein m}^{-2} \text{ s}^{-1}$  on a 12:12 hour light-dark cycle. At the end of this time period, growth potential was determined by examining the maximum standing crop of each flask.

Biomass was measured gravimetrically by filtering measured aliquots of culture through pre-weighed  $0.45 \mu\text{m}$  Millipore filters. The filters were then dried at  $100^{\circ} \text{C}$  for 24 hours, cooled, and weighed. Maximum standing crop for each treatment was calculated as  $\text{mg } \ell^{-1}$  and recorded. Complete details of the experimental design are outlined in Procedure 13 of Appendix I.

#### Vascular Plants

The macrophyte community of Lake of the Woods was sampled biweekly from May through August, 1981. Portions of the lake containing macrophytes were mapped and the area recorded. Samples were collected from each area using a white quadrat  $0.50 \text{ m} \times 0.50 \text{ m}$ . The quadrat was randomly thrown into the macrophyte beds and scuba divers then harvested by hand the above ground plant material within the quadrant. The vegetation was placed into a

mesh diving bag, loosely rinsed of sediments, and transferred to a plastic bag for storage on ice during transportation to the laboratory.

In the laboratory, the samples were thoroughly rinsed with water. They were allowed to drain overnight. The samples were sorted by species and placed into pre-weighed aluminum pans to determine wet weight. They were then placed in a forced-air drying oven for 48 hours at  $90-101^{\circ}\text{C}$  and weighed again for dry weight. The dried plant material was finely ground and subsamples were placed into pre-weighed containers for ignition at  $500-550^{\circ}\text{C}$  for one hour. After cooling in a dessicator, the ash weight was determined.

The measured weights were used to calculate macrophyte biomass in the lake on each sampling date. Total macrophyte biomass (wet, dry, and organic) in each portion of the lake was calculated based upon the sample weights and the surface area of that portion. The biomass values for all sections of the lake were summed and then divided by the total area of the lake supporting vascular plant growth to give a whole-lake, weighted average areal macrophyte biomass.

The concentrations of nitrogen and phosphorus in the macrophyte biomass were determined on each sampling date. Sunsamples of dried plant material were randomly selected for nutrient analyses. Phosphorus content was determined on persulfate digested samples (Wetzel and Likens 1979) using a molybdate-blue colorimetric method. The nitrogen content of the samples was determined using the persulfate digestion technique of Raveh and

Avnimelech (1979). The total nutrient pool incorporated in the macrophyte biomass was calculated using the measured nutrient concentrations and the measured macrophyte biomass values. Procedure 14 of Appendix I provides a detailed outline of the macrophyte nutrient analyses used in this study.

#### Zooplankton

The zooplankton community of Lake of the Woods was analyzed from March, 1981 through August, 1981. Vertical sample hauls were taken biweekly using a 30 cm diameter plankton net of 80  $\mu$  mesh (#20 size) at both the deep and shallow stations. The samples were preserved in a 5% formalin solution immediately after collection and placed on ice for transport to the laboratory.

In the laboratory, the samples were allowed to settle for three days and then decanted to a volume 500 ml. A random subsample of the 500 ml sample was obtained with a Stempel pipette and placed in a 5 ml Gannan counting tray for examination under a binocular microscope. Organisms were identified to species. Instar stages of copepods were also enumerated. Counts were converted to numbers per  $m^2$  by dividing by the area of the plankton net.

#### C. Sediment Analysis

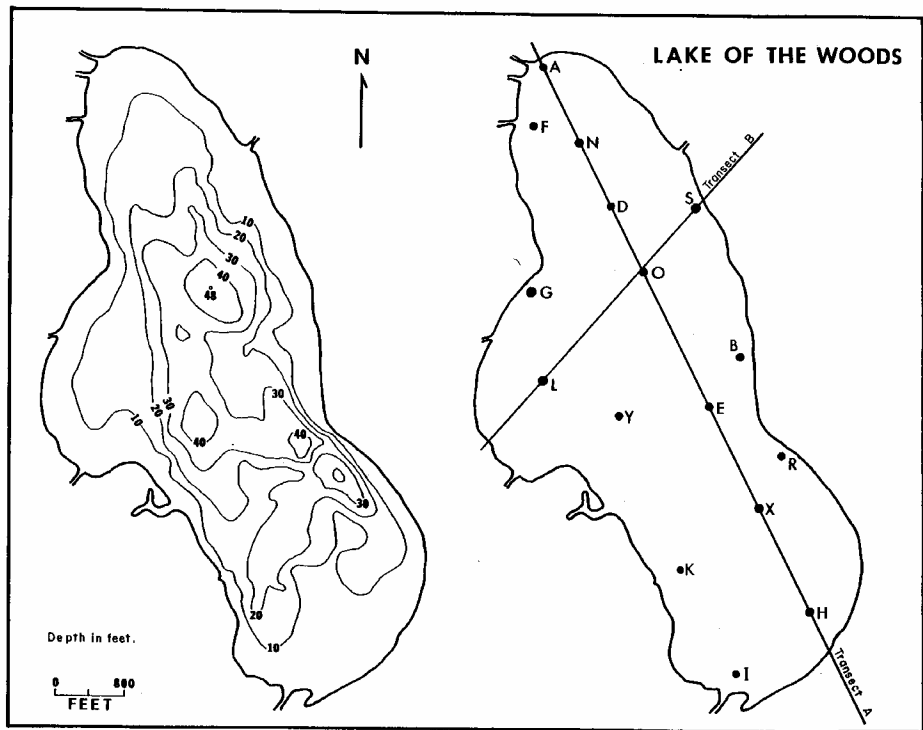
The sediments of Lake of the Woods were analyzed for their phosphorus content in July, 1981. Sixteen samples were collected along two main transects across the length of the lake (Fig. 5). These collection sites covered the major profundal zones of the lake and represented sediments at different depths. Duplicate samples were collected

at each site in acid-cleaned glass jars using scuba techniques. One jar was preserved with 10.8 N  $\text{H}_2\text{SO}_4$  for phosphorus analyses while the other jar was reserved for gravimetric analysis. Samples were stored on ice until their return to the laboratory.

The phosphorus content of the acidified samples was divided into two portions. The first of these, interstitial water phosphorus, was determined from a 50 ml sample of interstitial water which was digested using a persulfate technique. The second portion, acid-non-labile sediment bound phosphorus, was determined by performing a persulfate digestion on the dried sediments. Both portions used the molybdate-blue spectrophotometric procedure for measuring phosphorus. More complete details are presented in Procedure 15 of Appendix I.

The moisture, inorganic, and organic content of the sediment samples was determined gravimetrically. Thirty to forty  $\text{cm}^3$  of sediment were placed into pre-weighed crucibles and weighed to determine wet weight. The crucibles were then placed in a drying oven at  $101^\circ\text{C}$  for 24 hours and weighed again to determine dry weight. Finally, the samples were ashed at  $550^\circ\text{C}$  to determine the ash weight.

Fig. 5. Sediment sampling sites  
in Lake of the Woods.



#### IV. TRIBUTARY AND OUTLET MONITORING

##### A. Physical variables

###### Turbidity

The six tributary streams and outlet of Lake of the Woods showed considerable seasonal variations in turbidity (Appendix II; Fig. 1). Turbidity values of Sites 1 and 2 generally remained below 15 NTU for the study period except for late February and early June when values exceeded 20 NTU and 40 NTU, respectively. Tributaries 3 and 4 showed consistently large turbidity values throughout the sampling period as values generally remained above 25 NTU and often exceeded 50 NTU. Sites 5 and 6 showed the largest fluctuations in turbidity as values alternated from lows of less than 15 NTU to highs in excess of 50 NTU.

The outlet stream of Lake of the Woods had low turbidity values ( $< 10$ ) through mid-April. From late April to July, values increased to 45 NTU and then declined back to 15 NTU by August.

##### B. Chemical variables

###### Phosphorus

###### Total

There are significant differences in concentrations of total phosphorus (total P) among the tributaries of Lake of the Woods (Appendix II; Fig. 2). The concentrations of total P leaving the lake through the outlet remained fairly constant from December through August at 25-50  $\mu\text{g l}^{-1}$ . On a few sampling dates, however, total P concentrations exceeded 100  $\mu\text{g l}^{-1}$  in the outlet stream.

Tributaries 1 and 2 had the lowest total P concentrations of any of the inlet streams with values ranging from 25-50  $\mu\text{g } \ell^{-1}$  for December through June and values of 50-100  $\mu\text{g } \ell^{-1}$  for July and August. The remaining four tributaries all showed considerable fluctuations, ranging from 25-200  $\mu\text{g } \ell^{-1}$  or more. No apparent trends were obvious.

The differences in total P concentrations among tributary streams became more obvious when looking at the yearly weighted average total P concentrations. Tributaries 3, 4, and 5 had yearly weighted averages of 179, 133, and 132  $\mu\text{g } \ell^{-1}$ , respectively. Tributaries number 1, 2, and 6 had weighted averages of only 58, 48, and 70, respectively. The outlet stream had a yearly weighted average total P concentration of 53  $\mu\text{g } \ell^{-1}$ .

#### Soluble Reactive

Values of soluble reactive P (or "ortho" P) followed trends similar to total P values (Appendix II; Fig. 3). The outlet of Lake of the Woods (Site 7) had constant soluble reactive P values of less than 5  $\mu\text{g } \ell^{-1}$ . On two dates, late February and mid-March, concentrations of soluble reactive P in the outlet exceeded 20  $\mu\text{g } \ell^{-1}$ . No explanation can be offered for these two high values.

Soluble reactive P concentrations in the six tributaries fluctuated throughout the sampling period. Values were low < 30  $\mu\text{g } \ell^{-1}$  for the winter months. In late February, a sharp peak of soluble reactive P was noted in all six tributaries. Values exceeded 175  $\mu\text{g } \ell^{-1}$  at Sites 3 and 4 and ranged from 25-85  $\mu\text{g } \ell^{-1}$  among Sites 1, 2, 5, and 6. Following this strong pulse, all six streams showed a decline of soluble reactive P concentrations to base levels which remained fairly constant for the remainder of the sampling period. These base levels

were: 5-25  $\mu\text{g l}^{-1}$  at Site 1, 5-15  $\mu\text{g l}^{-1}$  at Site 2, 25-80  $\mu\text{g l}^{-1}$  at Site 3, 5-35  $\mu\text{g l}^{-1}$  at Site 4, 5-25  $\mu\text{g l}^{-1}$  at Site 5, and 1-15  $\mu\text{g l}^{-1}$  at Site 6. The large pulse of soluble reactive P occurring in all six tributary streams in late February is probably due to large runoff associated with snow melt at that time.

### Nitrogen

#### Ammonia

Ammonia concentrations for each stream are shown in Figure 4 of Appendix II. In the outlet of Lake of the Woods, ammonia concentrations varied seasonally. From December through mid-March, values averaged around 125  $\mu\text{g l}^{-1}$ , ranging from 75-160  $\mu\text{g l}^{-1}$ . During the spring and summer (April-August), ammonia concentrations remained below 60  $\mu\text{g l}^{-1}$  and occasionally dropped as low as 15  $\mu\text{g l}^{-1}$ . However, in early June a large pulse of ammonia in excess of 190  $\mu\text{g l}^{-1}$  was reported.

Ammonia concentrations in the six tributaries fluctuated considerably throughout the period from December through August. Tributary 1 had concentrations ranging from 25-60  $\mu\text{g l}^{-1}$  for most of this time, but exhibited a large pulse of ammonia in excess of 300  $\mu\text{g l}^{-1}$  in early June. The same pattern was evidenced by Tributary 2. Although Tributaries 3, 4, 5, and 6 showed much larger fluctuations throughout the year, they also exhibited very large pulses of ammonia in early June. This large June input of ammonia in all six tributaries may be linked to fertilization of farm fields; because of a very wet spring, the corn planting was delayed until late May and early June. It seems probable that the high ammonia values at this time are related to field runoff of fertilizer.

### Nitrite

Nitrite concentrations in the tributaries of Lake of the Woods remained low throughout the sampling period (Appendix II; Fig. 5). Nitrite comprised less than 5% of the annual total nitrogen loading at all sampling sites. Concentrations generally remained below  $75 \mu\text{g l}^{-1}$ , except during the summer months when higher values were noted on several dates. Site 3 had an annual weighted average nitrite concentration of  $79.2 \mu\text{g l}^{-1}$ , highest among all sampling locations. Sites 4, 5 and 6 had average annual concentrations between 50 and  $60 \mu\text{g l}^{-1}$ , while all other sites were much lower. Nitrite values, although high, are an insignificant fraction of the nitrogen entering and leaving the lake system.

### Nitrate

The nitrate concentrations in the tributaries of Lake of the Woods vary with time and location (Appendix II; Fig. 6). Average annual concentrations in all tributaries were extremely high, ranging from  $5.0 \text{ mg l}^{-1}$  at Site 4 to  $7.2 \text{ mg l}^{-1}$  at Site 3. Lower values were obtained in water leaving the lake at Site 7, averaging  $1.7 \text{ mg l}^{-1}$  and showing relatively little fluctuation throughout the year.

Although little spatial variation in nitrate levels of the tributaries was observed, distinct temporal trends were apparent. Values were highest in December, reaching a maximum of  $13 \text{ mg l}^{-1}$  at Site 1. By late January, concentrations had dropped in all streams to between  $3.1$  and  $6.8 \text{ mg l}^{-1}$ . Levels of nitrate at all sites fluctuated around  $6 \text{ mg l}^{-1}$  throughout the early summer, and then dropped to very low values of less than  $1.5 \text{ mg l}^{-1}$  at all sites

in mid-May. Reasons for this drastic decrease in nitrate concentrations are unknown at this time. By early June, values returned to previous levels with all sites exceeding  $6 \text{ mg } \ell^{-1}$ . Sites 4, 5 and 6 showed declining trends throughout July and August, while the other locations remained relatively constant.

#### Organic N

We had technical problems with the analysis of organic nitrogen during the early portion of this study. We originally began analyzing organic nitrogen using the micro-Kjeldahl technique. However, we were unable to obtain satisfactory precision with this technique and after consultation with EPA officials in Cincinnati and university chemists, we switched to a persulfate digestion technique described by Raveh and Avnimelech (1979). This technique was perfected and began producing reliable results beginning in February.

Extremely high concentrations of organic nitrogen were found in all six inlets, ranging up to  $36.1 \text{ mg } \ell^{-1}$  at Site 2 (Appendix II; Fig. 7). Weighted annual average organic nitrogen concentration at each site were calculated as: Site 1,  $7.9 \text{ mg } \ell^{-1}$ ; Site 2,  $11.0 \text{ mg } \ell^{-1}$ ; Site 3,  $14.2 \text{ mg } \ell^{-1}$ ; Site 4,  $8.1 \text{ mg } \ell^{-1}$ ; Site 5,  $8.5 \text{ mg } \ell^{-1}$ ; Site 6,  $11.4 \text{ mg } \ell^{-1}$ ; and Site 7,  $5.0 \text{ mg } \ell^{-1}$ . Obviously, Site 3 had the highest concentration of organic nitrogen.

All of the inlets showed similar temporal trends in organic nitrogen concentration. In February and March, values remained less than  $15 \text{ mg } \ell^{-1}$  in all cases. In April, all tributaries exhibited large increases in organic nitrogen, representing up to a seven-fold increase from March values. Sites 2 and 3 had organic nitrogen levels

in excess of  $34 \text{ mg } \ell^{-1}$  at this time. Concentrations declined in May and June, and remained less than  $16 \text{ mg } \ell^{-1}$  at all sites for the remainder of the study period. The outlet, Site 7, showed little temporal fluctuation in the concentration of organic nitrogen. Values consistently remained near  $5 \text{ mg } \ell^{-1}$  except for one date in June.

#### Silica

Silica concentrations in the tributaries of Lake of the Woods show similar patterns throughout the year (Appendix II; Figure 8). Values were the highest ( $4\text{--}6 \text{ mg } \ell^{-1}$ ) in December and January, but declined to their lowest levels ( $0.2\text{--}0.4 \text{ mg } \ell^{-1}$ ) in late February. Following this decline, values remained more or less stable at values averaging from  $3\text{--}4 \text{ mg } \ell^{-1}$ .

Silica concentrations in the outlet remained constant throughout the year at a level less than any of the inlet streams. Except for one sampling period in December, silica concentrations remained below  $2.0 \text{ mg } \ell^{-1}$  on all sampling dates.

#### Residue

##### Total

The amount of total residue in all six inlet streams of Lake of the Woods varied very little over the study period (Appendix II; Fig. 9). During most of the period, total residues were constant at  $0.5 \text{ g } \ell^{-1}$  at all sampling sites. However, in mid-February the values increased almost six fold to  $3.0 \text{ g } \ell^{-1}$  during spring snow melt and associated runoff. Concentrations of total residues returned to low levels by late February. No significant differences in total residues among the sampling sites was noted.

Concentrations of total residue in the outlet of Lake of the Woods remained very constant throughout the study period. Except for one date in December, values remained in the range of 0.25 - 0.45 g  $\ell^{-1}$  throughout the sampling period.

#### Organic

Levels of organic residues in the tributaries of Lake of the Woods remained very constant throughout the study period (Appendix II; Fig. 10). Concentrations of organic residues ranged from 0.10 to 0.25 g  $\ell^{-1}$ . No significant differences were noted among sampling sites, except at Sites 3 and 5 in late August when the organic residue concentration exceeded 0.5 g  $\ell^{-1}$ . No explanation for this increase is apparent. Concentrations in the outlet stream were very similar to the values found in the tributary streams, ranging from 0.10 - 0.45 g  $\ell^{-1}$ .

Organic residues accounted for about 50% of total residues at sampling sites for the major portion of the study period. However, during the high residue loads associated with spring snow melt, the amount of organic residues did not increase. This indicates that the heavy loading of residues to the stream during high runoff periods is mostly inorganic in nature.

#### Particulate Matter

##### Total

Total particulate matter in the six tributaries and single outlet of Lake of the Woods showed different trends (Appendix II; Fig. 11). Concentrations at Sites 1 and 2 remained below 15 mg  $\ell^{-1}$  for the entire study period, except for one date in January when the total particulate matter concentration reached > 50 mg  $\ell^{-1}$  at Site 1. Sites 3 and 4 showed some temporal variation in total particulate matter

values. Concentrations remained low ( $< 10 \text{ mg } \ell^{-1}$ ) from January through April, and then rose erratically to higher values ( $20\text{--}70 \text{ mg } \ell^{-1}$ ) for the remainder of the year. Concentrations of total particulate matter showed even more temporal variation at Sites 5 and 6. Values ranged widely from less than  $5 \text{ mg } \ell^{-1}$  to greater than  $60 \text{ mg } \ell^{-1}$ , with no discernible pattern.

Concentrations of total particulate matter in the outlet of Lake of the Woods were more consistent. They rose gradually from a low of  $1.0 \text{ mg } \ell^{-1}$  in January to a high of  $11.0 \text{ mg } \ell^{-1}$  in late August. One particular high value of  $23 \text{ mg } \ell^{-1}$  was reported in early June.

#### Organic

Particulate organic matter concentrations were less variable than the total particulate matter concentrations at all sampling sites (Appendix II; Fig. 12). In general, particulate organic matter comprised less than 25% of the total particulate matter on all dates.

#### Total Alkalinity

The total alkalinity concentrations of the tributaries of Lake of the Woods are shown in Figure 13 of Appendix II. There appears to be a net loss of alkalinity to the lake. The outlet of the lake, Site 7, has lower total alkalinity, about  $150 \text{ mg } \ell^{-1}$ , than the inlets.

Alkalinities of the six tributary streams show very similar temporal patterns. Values fluctuated between  $150\text{--}270 \text{ mg } \ell^{-1}$  at Sites 1 and 2, while at Sites 3 and 4 values were slightly less and fluctuated between  $100\text{--}270 \text{ mg } \ell^{-1}$ . Concentrations of total alkalinity at Sites 5 and 6 ranged from  $110\text{--}290 \text{ mg } \ell^{-1}$ . All six sites showed a large decrease in alkalinity in late February most likely associated with the high runoff from snow melt and rain at this time.

## V. LAKE MONITORING

### A. Physical variables

#### Temperature

Temperature fluctuations in Lake of the Woods show a pattern typical of northern latitude dimictic lakes (Appendix III; Figs. 1 a-c). Under ice cover in January, temperatures dropped to  $2.0^{\circ}\text{C}$  at the deep station and  $1.5^{\circ}\text{C}$  at the shallow station. Both sites showed an inverse stratification pattern of colder surface water typical of dimictic lakes in winter. By March, lake temperatures began rising again to  $7^{\circ}\text{C}$  as the lake began warming. During April and May the lake continued to warm. By early June, the lake had stratified with a thermocline beginning at 3.0 m. Bottom temperatures in the deep station remained around  $12^{\circ}\text{C}$  throughout the summer, while the surface temperatures climbed to  $22\text{--}26^{\circ}\text{C}$ . The shallow station also showed thermal stratification, but the bottom temperatures were much higher ( $19\text{--}22^{\circ}\text{C}$ ).

#### Conductivity

Conductivity measurements in Lake of the Woods show distinct differences between sampling dates as shown in Figures 2 a-d of Appendix III. At the deep station conductivity values averaged around  $540\text{--}565\ \mu\text{S cm}^{-1}$ . By mid-July, however, conductivity values showed sharp clinograde profiles. Conductivity was low at the surface ( $< 500\ \mu\text{S cm}^{-1}$ ) and increased dramatically ( $> 525\ \mu\text{S cm}^{-1}$ ) in the hypolimnion. This pattern also occurred at the shallow station.

### Turbidity

Turbidity profiles of Lake of the Woods show large differences throughout the study period (Appendix II; Figs. 3 a-c). Turbidity values in January were below 5 NTU at all depths. Similar values were reported in February with one exception. A very large plume of turbid water was observed flowing under the ice at the deep station and this yielded an incredibly large turbidity value of 25.0 NTU. In March and April, turbidity values increased to > 10 NTU at both stations. During this time, turbidity values increased with depth. Beginning in June, the upper 6 m of water had turbidity values in excess of 10 NTU. However, values in the lower depths were generally less than the upper waters. On one date, June 6, 1981, turbidity values at the surface reached levels greater than 30 NTU. On this same date, chlorophyll a values were the highest recorded indicating that the turbidity was due to large algal biomass.

### Secchi disk

The secchi disk depth in Lake of the Woods varied by almost three meters throughout the nine month study period (Appendix III; Fig. 4). In January, secchi disk was 2.7 m at the deep station and 2.2 at the shallow station. In February, the secchi disk depth at the deep station decreased to less than 0.5 m as a result of the turbid plume flowing under the ice. At this time, the shallow station had a secchi disk depth of 3.3 m. By late March, the secchi disk depth was 1.2 at both stations. From May through August, both the deep and shallow stations exhibited similar secchi disk values ranging from 0.5 - 1.3 m.

### Light attenuation

The total attenuation coefficients of the upper 3 m of Lake of the Woods ranged from about  $1.38 \text{ m}^{-1}$  to  $3.36 \text{ m}^{-1}$  (Appendix III; Table 1). The proportion of light attenuation due to chlorophyll a changed drastically throughout the summer. In early June, almost all of the light (91%) was attenuated by algae as evidenced by the low ratio of  $n_w/n$ . As the algal bloom in the lake decreased, so did the amount of light attenuated by algae. By late August, only 30% of the attenuation was due to algal pigments (i.e. -  $n_w/n = 0.70$ ).

### B. Chemical variables

#### pH

Values of pH in Lake of the woods showed variations with depth throughout the year (Appendix III; Figs. 5 a-c). Under ice cover in January and February, pH values ranged from 8.0 in the upper layers to 10.0 at the lower depths. Commencing in June, pH values showed sharp clinograde profiles with the upper 3 m remaining at pH = 8.0 while the lower layers had pH values of 7.0.

#### Dissolved oxygen

Dissolved oxygen readings for Lake of the Woods show some striking trends (Appendix III; Figs. 6 a-c). The mid-January oxygen curves for both the shallow and deep station showed a decline with depth. Values decreased from  $17.5 \text{ mg } \ell^{-1}$  at the surface to  $16.0 \text{ mg } \ell^{-1}$  at the shallow station and  $11.0 \text{ mg } \ell^{-1}$  at the deep station. This indicates that considerable heterotrophic activity was occurring in the lake at this time.

After spring turnover, oxygen concentrations remained uniform at about  $12.0 \text{ mg } \ell^{-1}$  (100 % saturation). By mid-April, however, a sharp

decline in oxygen concentrations was noted below 9.0 m. By June, oxygen depletion was noted at all depths below 3 m. Oxygen concentrations in the lower waters continued to decline, and by late June no oxygen existed below 6.0 m. This situation continued throughout the summer. Oxygen concentrations of the epilimnion during this time remained close to, or slightly above, 100% saturation.

### Phosphorus

#### Total

Total phosphorus concentrations in Lake of the Woods varied throughout the study period (Appendix III; Figs. 7 a-d). In January, total phosphorus concentrations ranged from  $25 \mu\text{g l}^{-1}$  at the surface to  $100 \mu\text{g l}^{-1}$  at 12 m. In February, a sharp peak of total phosphorus, ( $192 \mu\text{g l}^{-1}$ ) was noted at the surface; this is a reflection of the plume flowing under the ice at this time. With spring turnover in March, the total phosphorus profile was uniform with depth at  $60\text{--}65 \mu\text{g l}^{-1}$ . Profiles remained more or less uniform throughout May and June, ranging from  $25 - 50 \mu\text{g l}^{-1}$ . By July and August, however, sharp increases of total phosphorus in the anoxic hypolimnion were noted. Concentrations reached as high as  $183 \mu\text{g l}^{-1}$ .

#### Soluble Reactive

The amount of soluble reactive ("ortho") phosphorus in Lake of the Woods remained very small throughout the study months (Appendix III; Figs. 8 a-d). Only under the ice in January and February did concentrations exceed  $5.0 \mu\text{g l}^{-1}$  in the water column. On all other sampling dates, the concentrations of soluble reactive phosphorus were almost undetectable. Beginning in July, however, the concentrations of soluble reactive phosphorus at depths of 12 m or greater increased

consistently and reached values of  $40 \mu\text{g l}^{-1}$ .

### Nitrogen

#### Ammonia

Concentrations of ammonia in the water column of Lake of the Woods varied seasonally (Appendix III; Figs. 9 a-d). From January through March, ammonia concentrations were uniform with depth, averaging  $75 - 120 \mu\text{g l}^{-1}$ . In April, concentrations dropped to less than  $25 \mu\text{g l}^{-1}$ , but they were still fairly uniform with depth. Beginning in mid-May and continuing throughout the summer, clinograde ammonia profiles were found. Upper water (0-6 m) had concentrations ranging from  $10 - 50 \mu\text{g l}^{-1}$ . Below 3.0 m, ammonia concentrations increased with depth to maxima of several hundred  $\mu\text{g l}^{-1}$ . This is another reflection of the anoxic conditions existing in Lake of the Woods at depths greater than 3.0 m.

#### Nitrite

The nitrite concentrations in Lake of the Woods showed some moderate seasonal patterns (Appendix III; Figs. 10 a-d). From January through May, water column concentrations were uniform with depth and ranged from  $10 - 35 \mu\text{g l}^{-1}$ . From June through August, epilimnetic concentrations ranged from  $35 - 75 \mu\text{g l}^{-1}$ , while hypolimnetic concentrations ranged from  $65 - 200 \mu\text{g l}^{-1}$ . In mid-August hypolimnetic concentrations decreased dramatically in response to the anoxic hypolimnion.

#### Nitrate

The nitrate concentrations of Lake of the Woods followed classical seasonal trends (Appendix III; Figs. 11 a-d). From January through June, nitrate concentrations were uniform with depth, ranging from

0.5 - 2.5 mg  $\ell^{-1}$ . As the hypolimnion went anoxic from May through August, nitrate gradually disappeared from the lower depths of the lake. Epilimnetic concentrations remained close to 1.0 mg  $\ell^{-1}$  during this time. In August, nitrate concentrations in the epilimnion decreased sharply to less than 0.2 mg  $\ell^{-1}$ .

#### Organic

The organic nitrogen concentrations of Lake of the Woods were the largest nitrogen fraction in the lake, comprising over 50% of the total nitrogen on all sampling dates. Concentrations of organic nitrogen varied little with depth (Appendix III; Figs. 12 a-d). Concentrations remained stable at about 7 mg  $\ell^{-1}$  from January through May. In the months of June, July, and August concentrations of organic N averaged less than 7 mg  $\ell^{-1}$ .

#### Silica

Silica profiles in Lake of the Woods follow classical seasonal trends (Appendix III; Figs. 13 a-c). Concentrations were very low in January and remained below 1.0 mg  $\ell^{-1}$  through May as the diatom population grew and flourished. Beginning in June and continuing throughout the summer, the diatom population was replaced by other algae and concentrations of silica rose to values as high as 2.0 mg  $\ell^{-1}$ .

#### Residue

##### Total

Total residue profiles in Lake of the Woods show sharp differences throughout the sampling months (Appendix III, Figs. 14 a-d). Values remained below 0.1 g  $\ell^{-1}$  from January through March. In spring and summer, values were much higher ( $> 0.3$  g  $\ell^{-1}$ ). No obvious trends were noted.

### Organic

Organic residue profiles in Lake of the Woods also showed sharp seasonal differences (Appendix III; Figs. 15 a-d). Values averaged less than  $0.1 \text{ g } \ell^{-1}$  through March, but increased to almost  $0.2 \text{ g } \ell^{-1}$  the rest of the sampling period.

### Particulate Matter

#### Total

The amount of total particulate matter in Lake of the Woods remained only a small fraction (1-5%) of the total residue in the lake throughout the entire sampling period (Appendix III; Figs. 16 a-d). No seasonal patterns were noted from January through June and values averaged about  $10 \text{ mg } \ell^{-1}$ . In June, July and August, the values decreased to  $5 - 10 \text{ mg } \ell^{-1}$ . No distinct depth patterns were noted.

#### Organic

Profiles of particulate organic matter in Lake of the Woods were similar to the total particulate matter profiles (Appendix III; Figs. 17 a-d). In general, particulate organic matter represented in excess of 50% of the total particulate matter on any date indicating the presence of large phytoplankton populations in the lake.

### Alkalinity

The total alkalinity values of Lake of the Woods show seasonal changes that reflect the changes in the algal biomass of the lake (Appendix III; Figs. 18 a-d). From January through May, alkalinity profiles remain constant at about  $155 - 165 \text{ mg } \ell^{-1}$ . Beginning in June, the alkalinity of the epilimnion decreased to values as low as  $140 \text{ mg } \ell^{-1}$  and sharp clinograde profiles are seen. This is most likely due to the large blue-green algal populations found during this time period.

### C. Biological Variables

#### Bacteria

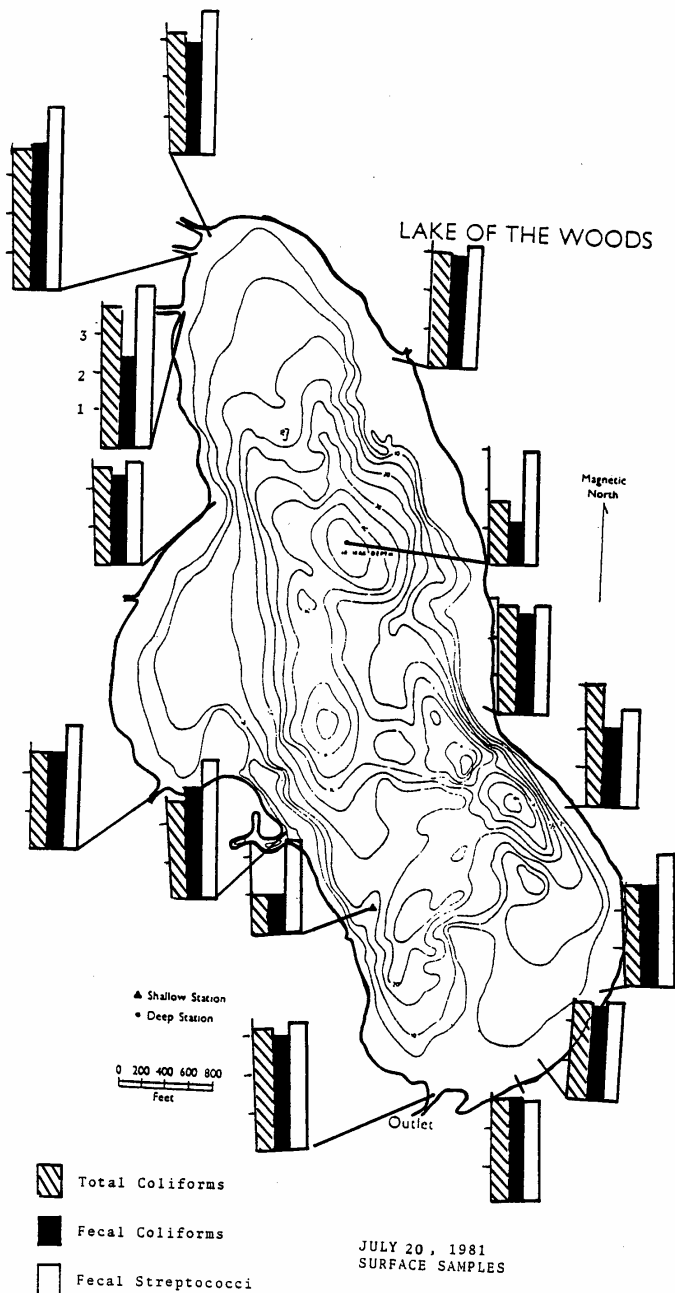
Results of the bacterial analyses of Lake of the Woods and its tributary waters show that the lake receives fecal contamination, and that this contamination impairs the water quality of the lake. Evidence also indicates that the fecal contamination is not only from animal sources but from human sources as well.

Total coliform counts in Lake of the Woods show that the in-lake waters generally have insignificant health hazards. Counts remained below 100 per 100 ml from June through August. However, total coliform levels exceeded 100 per 100 ml in May at the shallow station and in April and May at the deep station. Outflow water had counts in excess of 200 per 100 ml in May and July. Counts in the inlet ditches, however, were much higher, ranging upto 47,000 per ml. No consistent pattern of increasing or decreasing counts is seen in any of the tributaries as the summer progressed. All tributaries exhibited very high counts sometime during the sampling period (Appendix III; Table 2).

Fecal coliform counts for Lake of the Woods were low for in-lake samples on all dates. Counts in the outflow of the lake in July represent need for concern with verified values greater than 200. Fecal coliform counts in all of the tributaries exceeded the safety limit of 200 per 100 ml on almost all sampling dates. No pattern of fecal coliform contamination is evident in these streams. Fecal coliform counts are summarized in Table 3 of Appendix III.

Counts of fecal streptococci bacteria indicate a general increase during the warmer months (Appendix III; Table 4). The ratio of fecal coliform to fecal streptococci (FC:FS ratio) was used to determine the

Fig. 6. Bacteriological survey of Lake of the Woods on July 20, 1981. Histograms represent log numbers of bacteria.



source (i.e. animal or human) of the fecal pollution. Values approaching 3.0 represent human waste contamination while values less than 0.7 are indicative of livestock wastes. On several dates, values of the FC:FS ratio for Lake of the Woods exceeded 0.7 (Appendix III; Table 5). In April, tributaries #1 and #3 had FC:FS ratios greater than 3.0. This strongly indicates that human waste contamination is present in the lake or its watershed and can be of significant concern at times throughout the year.

Standard plate counts for Lake of the Woods show increasing values as the waters warmed (Appendix III; Table 6). In-lake counts were generally lower than tributary streams.

A survey around the lake taken on July 20, 1981, showed no large differences in bacterial counts among sites (Fig. 6). In general, the deep and shallow in-lake stations show counts approximately  $10^1$  less than the shoreline samples. The shoreline samples generally had high counts with fecal coliform often exceeding 200 per 100 ml. The pattern of generalized high counts on all shoreline samples indicate many sources of fecal pollution surrounding the lake as opposed to an isolated tributary or home.

In summary, microbiological data indicate excessive fecal pollution in the inlets to Lake of the Woods throughout much of the summer. This contamination occurs not only from animal wastes, but on occasion from human sources as well. Shoreline samples show reason for concern as fecal coliform counts often exceeded 200 per 100 ml. Samples taken further out in the lake show no health hazard.

### Chlorophyll a

Chlorophyll a values in Lake of the Woods showed significant seasonal trends (Appendix III; Figs. 19 a-d). In January and February, chlorophyll a values were less than  $5 \mu\text{g l}^{-1}$  with no vertical stratification. By late March, 1981, and ice-free conditions, the chlorophyll levels were up to  $40\text{--}50 \mu\text{g l}^{-1}$  as the phytoplankton began responding to increasing water temperatures and light intensities. Concentrations of chlorophyll a decreased to  $20 \mu\text{g l}^{-1}$  in mid-April and then increased to about  $40 \mu\text{g l}^{-1}$  by early May. The phytoplankton in Lake of the Woods continued to grow and by early June a chlorophyll maximum of over  $125 \mu\text{g l}^{-1}$  was noted in the upper 3.0 m of water. Chlorophyll a concentrations below 3.0m averaged only  $25 \mu\text{g l}^{-1}$  or less in response to the low light intensities at those depths. The huge peak of chlorophyll declined in late June and epilimnetic concentrations remained in the range of  $25\text{--}45 \mu\text{g l}^{-1}$ . Hypolimnetic concentrations during the summer months were very low ( $< 15 \mu\text{g l}^{-1}$ ).

### Phytoplankton communities

Lake of the Woods exhibited a well-defined pattern of seasonal succession. Following the fall overturn, the phytoplankton community was dominated by diatoms. During the winter months, blue-green and green algae, diatoms, and cryptomonads were the dominant species. During spring overturn, the phytoplankton community once again exhibited a diatom pulse. Cryophytes and cryptomonads were also more numerous during spring overturn. Blue-green algal species replaced diatoms and cryptomonads as spring progressed and dominated the phytoplankton community throughout the summer months.

In late January, 1981, the phytoplankton community was dominated by the cyanophyte, Oscillatoria sp., the chlorophyte, Ankistrodesmus falcatus, and the diatoms, Asterionella formosa and Stephanodiscus niagarae (Appendix III; Fig. 20a). Oscillatoria comprised the greatest percentage by both volume and number per ml in the upper depths, but decreased as diatom biomass increased with depth as a function of settling out under ice cover. Algal cell numbers per ml were at their annual minimum of ca. 120 cells  $\text{ml}^{-1}$  due to low temperature and light. Average total cell volume was ca.  $1.3 \times 10^6 \mu\text{m}^3 \text{ ml}^{-1}$ , also very close to the annual minimum. Chlorophyll a concentrations were correspondingly low, averaging ca.  $2.5 \mu\text{g l}^{-1}$  at both shallow and deep stations.

By mid-February, the diatoms had almost completely settled out and Oscillatoria and Cryptomonas became the dominant species (Appendix III; Fig. 20b). Oscillatoria comprised ca. 80% of the total volume at all depths, while Cryptomonas accounted for ca. 80% of the total numbers  $\text{ml}^{-1}$  in the epilimnion. This small flagellate was concentrated in the upper depths and decreased with depth in response to reduced light penetration.

Reflecting this, average total cell numbers  $\text{ml}^{-1}$  from 0.0 to 4.5 m at the shallow and deep station were ca. 350 cells  $\text{ml}^{-1}$ . In the lower depths (6-12 m) of the deep station numbers averaged ca. 135 cells  $\text{ml}^{-1}$ . Average chlorophyll a concentration at the shallow station was  $2.8 \mu\text{g l}^{-1}$  and  $4.0 \mu\text{g l}^{-1}$  at the deep station.

During spring overturn in late March - early April, the algal biomass in Lake of the Woods increased in response to increases in temperature, light, and nutrient concentrations. The resulting diatom

pulse consisted of Stephanodiscus tenuis, Stephanodiscus niagarae, Asterionella formosa, and Diatoma sp. The chrysophyte, Dinobryon sociale, also bloomed during overturn.

By the end of March, average volume  $\text{ml}^{-1}$  had increased to ca.  $1.8 \times 10^7 \mu\text{m}^3 \text{ml}^{-1}$  and average numbers  $\text{ml}^{-1}$  had increased to ca. 9,300 cells  $\text{ml}^{-1}$  (Appendix III; Fig. 20c). Average chlorophyll a concentration increased accordingly to ca.  $46 \mu\text{g l}^{-1}$ . There were no significant differences observed between deep and shallow stations.

By April 13, blue-green algal biomass had increased and diatom biomass decreased in response to an increase in temperature and a decrease in light penetration. The dominant blue-green species were Oscillatoria and Chroococcus spp. and the dominant diatoms were Asterionella formosa, Stephanodiscus tenuis, Stephanodiscus niagarae, and Fragilaria crotonensis. Blue-greens accounted for ca. 60% of the total volume, while diatoms comprised ca. 50% of the total number. Cryptomonas sp. accounted for ca. 15% of the total number, but due to their small size, had no significant contribution to total volume. Average total volume was  $1.0 \times 10^7 \mu\text{m}^3 \text{ml}^{-1}$  and average total numbers  $\text{ml}^{-1}$  was ca. 3,600. Average chlorophyll a concentration was  $21 \mu\text{g l}^{-1}$ . No significant differences between shallow and deep stations were observed (Appendix III; Fig. 20d).

In May, as stratification began to develop and water temperatures increased, blue-greens continued to increase in dominance, while diatoms decreased (Appendix III; Fig. 20e). Average total volume was  $3.7 \times 10^7 \mu\text{m}^3 \text{ml}^{-1}$  and average total number was ca. 4,000 cells  $\text{ml}^{-1}$ . Average chlorophyll a concentration increased to  $32 \mu\text{g l}^{-1}$  in response to the greater density of Oscillatoria.

Later in May, as stratification continued to develop, densities of blue-green algae increased further and diatoms continued to settle out. Oscillatoria, Chroococcus, Aphanizomenon sp. and Microcystis aeruginosa were the dominant species comprising over 90% of both total volume and total number in the epilimnion (Appendix III; Fig. 20f). The biomass of diatoms increased as they settled out in the hypolimnion. In response to higher water temperature at the shallow station, average total numbers were  $11,000 \text{ cells ml}^{-1}$  and average total volume was  $9.6 \times 10^7 \mu\text{m}^3 \text{ ml}^{-1}$ . Likewise, warmer water from 0 - 4.5 m at the deep station resulted in  $9,000 \text{ cells ml}^{-1}$ , with an average volume of  $8.0 \times 10^7 \mu\text{m}^3 \text{ ml}^{-1}$ , and an average chlorophyll a concentration of  $c. 77 \mu\text{g l}^{-1}$ . Average total number from 6 to 9 m at the deep station decreased to  $2,700 \text{ cells ml}^{-1}$  while average total volume decreased to  $ca. 2.95 \times 10^7 \mu\text{m}^3 \text{ ml}^{-1}$ .

By June 1, stratification had been completely established and community composition remained relatively unchanged the remainder of the summer. The dominant blue-greens, Oscillatoria, Aphanizomenon, Microcystis, and Chroococcus accounted for over 90% of the numbers and volumes in both the epilimnion and metalimnion, while the senescent diatoms from spring overturn continued to sink down to the hypolimnion (Appendix III; Fig. 20g).

Due to possible thermal and photoinhibition at the shallow station, average total numbers decreased to  $6,500 \text{ cells ml}^{-1}$  and average total volume decreased to  $5.2 \times 10^7 \mu\text{m}^3 \text{ ml}^{-1}$ . Biomass estimates at the deep station, however, were at the annual maximum. From 0 - 4.5 m, average total number was  $26,150 \text{ cells ml}^{-1}$ , composed almost entirely of Oscillatoria, average total volume was  $2.24 \times 10^8 \mu\text{m}^3 \text{ ml}^{-1}$ , and

average chlorophyll a concentration was ca.  $131 \mu\text{g l}^{-1}$ . Due probably to low light, average total numbers and average total volume from 6-12 m at the deep station were  $950 \text{ cells ml}^{-1}$  and  $8.7 \times 10^6 \mu\text{m}^3 \text{ ml}^{-1}$ , respectively. Average chlorophyll a concentration was only  $11 \mu\text{g l}^{-1}$  at this time in deep water.

By the end of June biomass declined, possibly in response to nutrient depletion. At both the shallow station and 0 - 4.5 interval of the deep station average total numbers were  $6,650 \text{ cells ml}^{-1}$ , average total volume was  $4.6 \times 10^7 \mu\text{m}^3 \text{ ml}^{-1}$ , and average chlorophyll a concentration was ca.  $51 \mu\text{g l}^{-1}$ . Again, biomass estimates from 6-9 m were much lower due to self-shading and the completion of diatom settling (Appendix III; Fig. 20h). Community diversity declined even further as just a few species of green algae were represented along with the dominant blue-green species.

By early July, photoplankton biomass had declined further, but seemed to stabilize at a level which persisted through August. During these two months, at the shallow station and from 0 - 4.5 m at the deep station, average total number fluctuated around  $4,000 \text{ cells ml}^{-1}$  while average total volume remained at ca.  $2.5 \times 10^7 \mu\text{m}^3 \text{ ml}^{-1}$ . Chlorophyll a concentration stabilized at  $35 \mu\text{g l}^{-1}$ . Again in the lower depths of the deep station, biomass indices remained much lower than the surface waters due to reduced light penetration. Community composition also remained relatively unchanged throughout July and August (Appendix III; Figs. 20h, i, j).

The change in algal biomass during the study period is shown in Figures 21a and 21b of Appendix III. The average number of cells  $\text{ml}^{-1}$

ranged from a low of only a 100 or so per ml in late January to a high of over 26,000 per ml in early June. Changes in algal volumes exhibited similar trends. No obvious differences were noted between the deep and shallow stations, but during the summer months algal biomass decreased below 4.5 m at the deep station.

#### Primary productivity

Integral photosynthesis in Lake of the Woods totaled  $262.5 \text{ mg C m}^{-2} \text{ h}^{-1}$  on June 4. Primary productivity was greatest at the surface and declined with depth. Gross photosynthesis ceased at approximately 2 m, coinciding with the 1% surface light level. No surface photoinhibition was apparent.

On July 23, integral photosynthesis totaled  $279.4 \text{ mg C m}^{-2} \text{ h}^{-1}$ . Gross photosynthesis was greatest at the surface, decreased at 1.5 m due to self-shading, then exhibited a slight bulge at 2.0 m, possibly due to actively-photosynthesizing, low-light adapted Oscillatoria. Photosynthesis ceased at approximately 3.5 m, slightly below the 1% surface light level.

On August 20, integral photosynthesis totaled  $182.66 \text{ mg C m}^{-2} \text{ h}^{-1}$ . A maximum of  $100.9 \text{ mg C m}^{-3} \text{ h}^{-1}$  was obtained at 1.0 m. Gross photosynthesis ceased at 3.0 m or about the 1% surface light level. Surface photoinhibition was apparent on this date. Primary productivity profiles for all three dates are presented in Figure 22 of Appendix III.

#### Algal assay for nutrient limitation

Results of the E.P.A. A.A./B.T. (Algal Assay/Bottle Test) clearly indicate that phytoplankton growth in Lake of the Woods during the summer months is phosphorus-limited (Appendix III; Fig. 23). Addition of  $0.05 \text{ mg P l}^{-1}$ , singly and in combination with  $1.00 \text{ mg N l}^{-1}$ ,

stimulated a maximum standing crop of approximately fourteen times that of the lake water control. The differential response to N and P additions is reflected by the ambient surface nutrient concentrations in the test lake water. The ratio of inorganic N to soluble reactive P in the test lake water was ca. 1100:1, indicating an extremely phosphorus limited condition. The total N to total P ratio of ca. 65:1 also indicated extreme phosphorus limitation.

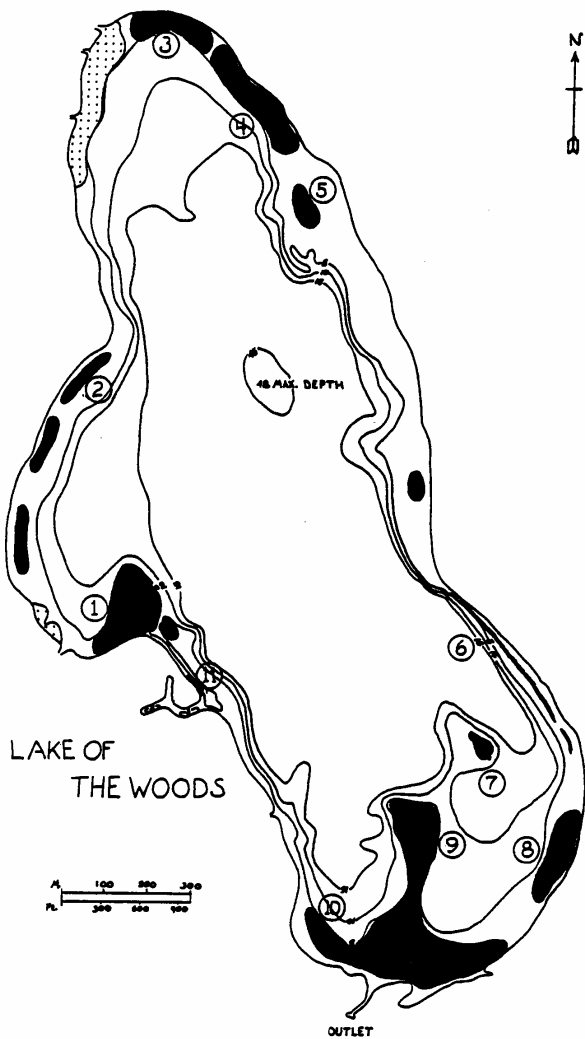
#### Vascular plants

The macrophyte beds in Lake of the Woods cover 9.1% of the lake surface (Fig. 7). The greatest densities occurred at sampling sites 1, 3, and 4. The average water depth of the sampling sites was 1.5 m.

Changes in the total lake biomass of aquatic macrophytes throughout the May - August sampling period are shown in Figure 24 of Appendix III. Initial biomass in May totaled  $4.02 \times 10^4$  kg dry weight. Biomass peaked in early June at  $25.4 \times 10^4$  kg dry weight and then declined through August. The ratio of dry weight to wet weight of the macrophytes ranged from 0.08 to 0.15 throughout the season. Areal biomass values ranged from 46 to 293 g dry weight  $m^{-2}$  during the sampling period.

There are few macrophyte species found in Lake of the Woods. The dominant plant in most areas of the lake is Myriophyllum spicatum, the Eurasian mil-foil. The only other species noted were Nuphar advena, Potamogeton crispus, and Potamogeton sp. and these were restricted to specialized locations in the lake. They grew in open water only in late summer as their competitor, M. spicatum, declined. For all

Fig. 7. Distribution of macrophytes in Lake of the Woods.  
Numbers indicate sampling sites.



practical purposes, the aquatic macrophyte population of Lake of the Woods is a monoculture of M. spicatum.

The nitrogen and phosphorus content of the macrophyte tissue was examined in detail. The amount of phosphorus in the tissue of M. spicatum remained in the range of 1.0 - 1.4 mg P (g dry weight)<sup>-1</sup> throughout the season. This critical P concentration does not fall below the minimum value of 0.08% for M. spicatum; this may be why M. spicatum is competitively dominant in Lake of the Woods. Nitrogen content of M. spicatum was higher ranging from 14 - 24 mg N (g dry weight)<sup>-1</sup>.

The total amount of phosphorus tied up in macrophyte tissue of the lake ranged from 6 - 41 kg, while the total amount of nitrogen in macrophyte biomass ranged from 95 - 600 kg. Maximum nutrient pools were found in June, and decreased as macrophyte biomass decreased (Appendix III; Fig. 25).

#### Zooplankton

During the period of spring overturn, and concurrent with spring algal pulse, the zooplankton community experienced large population increases. The dominant spring rotifer was Keratella cochlearis. In late May, as the lake thermally stratified, K. cochlearis remained dominant but there was a significant increase in Polyarthra dolicoptera numbers. The populations of K. cochlearis and Polyarthra dolicoptera persisted throughout the summer months.

Daphnia galeata mendotae and Bosmina longirostris were the dominant cladocern species throughout the entire study period.

Two species of cyclopoid copepods were dominant in Lake of the Woods during the study months. The winter species was Diacyclops

thomasi, which maintained dominance from fall overturn through mid-spring. During the onset of thermal stratification, Mesocyclops edax replaced D. thomasi as the major cyclopoid taxa, and it remained dominant until the beginning of September.

Skistodiaptomus oregonensis was the most abundant calanoid copepod species throughout the entire study period. Calanoids were generally less numerous than cyclopoids during virtually all sampling dates. Fluctuating population sizes of S. oregonensis during the summer months may in part be due to predation by fish and macroinvertebrate predators. Table 8 of Appendix III lists the major zooplankton species found in Lake of the Woods during the study period.

## VI. SEDIMENT ANALYSIS

Superficial sediment and interstitial water samples were collected in July using SCUBA techniques at 16 sites in Lake of the Woods (Fig. 5).

Two basic sediment types were identified on the basis of textural qualities: fine organic silt and small gravel-sand. Fine organic silt (Type 1) was the most abundant sediment type in the lake basin. Dry weight, moisture and organic content of each sediment type are given in Table 9 of Appendix III.

Phosphorus concentrations in the interstitial water of Lake of the Woods were positively correlated with increasing depth and reached a maximum  $3363 \mu\text{g P l}^{-1}$  at 12.0 m (station E). This correlation was evident at all locations sampled within the lake basin. The relationship of interstitial water phosphorus content (y) to depth (x) was described by the linear equation:

$$y = 188.54 (x) + 71.26 \quad 1)$$

Linear regression analysis indicated that depth explained over 81% of the variability in the data points ( $r^2 = 0.814$ ). Table 10 of Appendix III presents the phosphorus concentration of interstitial water at each station in Lake of the Woods.

Acid-nonlabile, sediment-bound phosphorus concentrations exhibited a strong positive correlation with sample depth. Linear regression analysis of sediment-bound phosphorus content (y) against depth (x) revealed that depth explained about 88% of the variability in the data points ( $r^2 = 0.879$ ). This relationship is described by the linear equation:

$$y = 71.56 (x) + 104.99 \quad 2)$$

where

y = sediment-bound phosphorus concentration  
of sediment

x = sample depth

This regression analysis included all samples except A and X. Sample A was collected in water 1.0 m deep directly adjacent to the inflow of tributary #5. The high phosphorus content found here ( $609.1 \mu\text{g P (g dry weight sediment)}^{-1}$ ) is probably due to the close proximity of this station to tributary sources transporting substantial amounts of phosphorus into the lake. In contrast, sample X was collected at 4.0 m and had a low phosphorus concentration of  $94.0 \mu\text{g P (g dry weight sediment)}^{-1}$ . This can be attributed to the sediment type at this location. The small gravel-sand substrate (Type 2) has a much lower surface area to volume ratio than the normal sediment type found in Lake of the Woods (Type 1). It would therefore be expected to have less available binding sites for phosphorus, yielding an overall reduction in the phosphorus content per unit sediment. Aside from these samples, phosphorus content ranged from 76.6 to  $1053.4 \mu\text{g P (g dry weight)}^{-1}$  of sediment. Table 11 of Appendix III provides complete data on acid-nonlabile, sediment-bound phosphorus content at each station.

Calculation of the lake sediment phosphorus pool required expression of total phosphorus content per unit volume of sediment. One square meter of sediment surface by 10 cm deep was the unit selected since studies have shown the top 10 cm to be the actively mixed sediment zone (Naumann 1930; Hayes 1964; Lee 1970). Phosphorus trapped in lower layers is

essentially "locked-up" and made unavailable to the overlying water. Hence, we have designated the term "unit sediment" to refer to a volume of sediment one  $m^2$  by 10 cm deep.

The volume of water and mass (dry weight) of sediment included in the 100 cm x 100 cm x 10 cm sediment unit was calculated for each of the two sediment types by multiplying the values listed in Table 9 of Appendix III by  $10^5$ . Fine organic silt, Type 1, contained 90.60 l of water and 14.31 kg dry weight per sediment unit. Small gravel-sand (Type 2) contained 46.23 l of water and 126.04 kg dry weight per sediment unit.

The amount of phosphorus tied up in the interstitial water per sediment unit was calculated by:

$$S_I = \frac{B \times C}{1000} \quad 3)$$

where

$S_I$  = total phosphorus content (mg P  $(10^5 \text{ cm}^3)^{-1}$ )

B = phosphorus concentration in interstitial water ( $\mu\text{g l}^{-1}$ )

C = total water content {l  $(10^5 \text{ cm}^3)^{-1}$ }

Acid-nonlabile sediment-bound phosphorus was calculated in a similar fashion:

$$S_B = D \times E \quad 4)$$

where

$S_B$  = sediment-bound total phosphorus {mg P  $(10^5 \text{ cm}^3)^{-1}$ }

D = phosphorus content of sediment {mg P (kg DW of sediment) $^{-1}$ }

E = total dry weight content (kg DW sediment)

Determination of the total phosphorus pool in the sediments of Lake of the Woods was accomplished by calculating the amount of phosphorus in each five foot depth interval and summing these values. The total phosphorus content per unit sediment,  $S_t$  {g P (10<sup>5</sup> cm<sup>3</sup>)<sup>-1</sup>}, in each five foot interval was calculated as follows: 1) multiple data points in any five foot interval were averaged to determine the mean, or 2) values for unsampled strata were estimated by averaging the values of  $S_t$  calculated for the intervals immediately preceding and following the unknown stratum. The total phosphorus in each stratum as g P was determined by multiplying the value of  $S_t$  {g (10<sup>5</sup> cm<sup>3</sup>)<sup>-1</sup>} by the number of sediment units (10<sup>5</sup> cm<sup>3</sup>) in that interval. Calculations of the number of sediment units in each stratum were difficult due to the irregular geometric configuration of the lake basin. Numerical analysis was employed to estimate the proportion of sediment area in each depth stratum. The surface area of saggital sections through the lake at each five foot depth interval was determined planimetrically from the bathymetric map (Appendix III, Table 12). Each isobath was then mathematically converted to a circle of equivalent area. The resultant series of circles was oriented according to depth along a perpendicular axis running through their centers, forming a cone with a base equal to the surface area of the lake and a height equal to the maximum depth. The cone was sectioned into frustra based on depth, and the surface area of each frustrum was approximated by using this equation:

$$2\pi \int_0^5 x \sqrt{1 + \left(\frac{dx}{dy}\right)^2} \cdot dy \quad 5)$$

The sediment surface area of each depth interval was summed to yield total surface area of the entire lake basin. Using the percentages obtained through numerical analysis, the total area of sediment surface in each depth stratum could then be calculated using:

$$A_x = \phi \cdot A_t \quad 6)$$

where

$A_x$  = basin surface area in each 5 ft. stratum ( $m^2$ )

$\phi$  = relative % of  $A_t$  in stratum X

$A_t$  = total lake basin area ( $1.9715 \times 10^6 m^2$ )

The total amount of phosphorus contained in the sediments of a given depth interval was thus calculated as:

$$P_x = \frac{A_x \cdot S_t}{1000} \quad 7)$$

where

$P_x$  = sediment phosphorus pool in interval X (g)

$A_x$  = sediment units in interval X ( $m^2$ )

$S_t$  = unit P content for interval X {mg P (unit sediment) $^{-1}$ }

The total phosphorus pool for each depth stratum is given in Table 12 of Appendix III.

Data indicate that most of the phosphorus is accumulating in the deepest part of the basin. Sediments deeper than 20 ft. constitute about one-third of the total surface area of the basin, but contain approximately one-half of the total phosphorus pool. This situation results from sediment focusing, and is typical of many glacial lakes in the area.

Results of the phosphorus budget (see Section VIII A) indicate that approximately 72% of the phosphorus entering the lake is retained

in the system. Virtually all of this eventually reaches the sediments where it accumulates. The total phosphorus pool contained in the superficial sediments of Lake of the Woods is 16,393 kg. This amount is 16.2 times the measured annual phosphorus input, and constitutes 90.8% of all of the phosphorus present in the entire lake system (Table 4).

The morphometry of the lake basin and resultant spatial distribution of the sediments are important considerations when examining the water quality of Lake of the Woods. Over 42% of the surface area of the lake basin is in ten feet of water or less. These shallow sediments are subject to resuspension and mechanical agitation from wind mixing, boating, and swimming. This mixing often causes increased turbidity and nutrient release from the sediments, and ultimately contributes to water quality problems.

## VII. HYDROLOGIC BUDGET

The hydrologic budget of Lake of the Woods was calculated from measurements of water movement and precipitation through the lake system over the nine month sampling period. The basic hydrologic model used was:

$$V + Q + P + R - O - E + G \quad 1)$$

where

V = change in lake volume

Q = tributary inputs

P = precipitation on the lake surface

R = direct runoff

O = outlet

E = evapotranspiration

G = groundwater

The change in lake volume, V, was assumed to be zero since no measurable change in lake height was noted on a yearly basis.

The instantaneous volume flow of each inlet and outlet stream was calculated using empirically determined discharge curves. These curves were determined by fitting an equation of the form

$$H = \frac{H_{\max} F}{F + k} \quad 2)$$

where

H = water level height (cm)

$H_{\max}$  = maximum water level height (cm)

F = discharge ( $\text{m}^3 \text{s}^{-1}$ )

k = half saturation constant for flow

to plots of measured stream flow ( $\text{m}^3 \text{s}^{-1}$ ) versus water level height (cm).

Best fit parameters  $H_{\max}$  and  $k$  were obtained using a simplex optimization procedure (Nelder and Mead 1965; Deming and Morgan 1973; Morgan and Deming 1974; and King et al. 1975). Figures 1-7 of Appendix IV show the discharge curves for the tributary streams of Lake of the Woods.

Precipitation was measured by installing rain gauges at three locations around the lake. However, vast discrepancies in the data collected by local residents made this information useless. Alternatively, precipitation data were supplied by the National Weather Service station at South Bend (Appendix IV; Table 1). Due to the close proximity of Lake of the Woods to South Bend and the uniformity of physiographic and climatic features in the region, this data was assumed to be representative of ambient precipitation conditions at the lake.

Evapotranspiration was estimated using the data of Reussow and Rohne (1975). Studies have shown that values for evapotranspiration do not deviate significantly from such published data.

No direct measures of groundwater flux or direct runoff were made. These two hydrologic categories were grouped together and solved by difference from equation 1.

Calculation of total water input to Lake of the Woods was done indirectly. Infrequent measurements of water height in the individual streams made computation of daily, or even weekly, discharge volumes impossible. Instead, the following procedure was used. The amount of runoff coming into Lake of the Woods was assumed to be proportional to the amount of runoff entering another lake, Lake Waubesa, studied during this time period. Measurements of stream runoff to Lake Waubesa were quite precise and the total precipitation to the two lake water-

sheds was identical. Hence, total stream runoff to Lake of the Woods was found as:

$$\frac{A_d(\text{LOW})}{A_d(\text{W})} \times \text{Stream runoff (W)} = \quad 3)$$

$$\frac{24.5005}{37.4348} \times 7.491 \times 10^6 \text{ m}^3 = 4.902 \times 10^6 \text{ m}^3$$

where

$A_d(\text{LOW})$  = watershed area of Lake of the Woods

$A_d(\text{W})$  = watershed area of Lake Waubesa

Stream runoff (W) = stream runoff to Lake Waubesa

The percentage of the total stream input carried by each individual stream was determined from measurements of daily streamflow on dates when complete information was available.

The yearly hydrologic budget of Lake of the Woods is summarized in Table 2. A total water input, Q, of  $5.691 \times 10^6 \text{ m}^3$  entered Lake of the Woods during the 12 month period. Tributary #4 was the major input, contributing  $1.716 \times 10^6 \text{ m}^3 \text{ y}^{-1}$  or 25.6% of the yearly water. Tributary #5 followed closely with  $1.387 \times 10^6 \text{ m}^3 \text{ y}^{-1}$  or 20.7% of the water. Tributary #1 contributed 11.9% or  $7.992 \times 10^5 \text{ m}^3 \text{ y}^{-1}$ , while Tributary #3 added  $7.011 \times 10^5 \text{ m}^3 \text{ y}^{-1}$  (10.5%). Tributary #2 added only 2.5% ( $1.667 \times 10^5 \text{ m}^3 \text{ y}^{-1}$ ) and tributary #6 added 2.1% ( $1.373 \times 10^5 \text{ m}^3 \text{ y}^{-1}$ ). Direct lake precipitation of  $1.790 \times 10^6 \text{ m}^3 \text{ y}^{-1}$  accounted for the remaining 26.7%.

Evapotranspiration losses from Lake of the Woods were estimated at  $1.002 \times 10^6 \text{ m}^3 \text{ y}^{-1}$  based upon a unit loss of  $60.96 \text{ cm m}^{-2} \text{ y}^{-1}$ . Since lake storage was constant, the outlet was assumed to carry the remaining water leaving the system. This amounted to  $5.691 \times 10^6 \text{ m}^3 \text{ y}^{-1}$  or 85% of the water losses from Lake of the Woods. Figure 8 shows the relative hydrologic contribution of each source for Lake of the Woods.

The hydraulic residence time of water in the lake,  $t_w$ , was calculated according to the formula

$$t_w = \frac{V}{Q} \quad 4)$$

where

$V$  = lake volume ( $m^3$ )

$Q$  = mass water loading ( $m^3 y^{-1}$ )

For Lake of the Woods this becomes

$$t_w = \frac{7.848 \times 10^6}{6.692 \times 10^6} = 1.17$$

Thus, it takes 1.17 y for the water volume of the lake to be replaced.

This relatively short residence time greatly affects the response of the lake to a given nutrient load.

Areal water loading,  $q_s$  ( $m y^{-1}$ ), for Lake of the Woods is given by

$$q_s = \frac{Q}{A_o}$$

where

$Q$  = mass water loading ( $m^3 y^{-1}$ )

$A_o$  = surface area of the lake ( $m^2$ )

Hence,

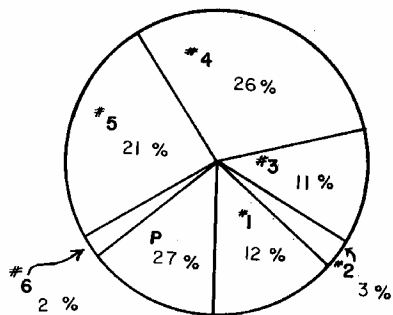
$$q_s = \frac{6.692 \times 10^6 m^3 y^{-1}}{1.643 \times 10^6 m^2} = 4.07 m y^{-1}$$

This areal water loading value is useful for modelling nutrient concentrations in the lake.

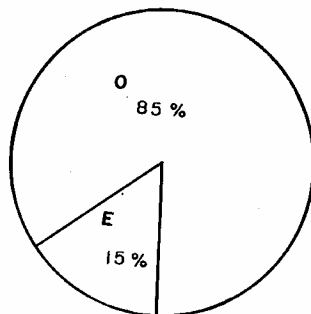
Fig. 8. Hydrologic budget for Lake of the Woods.  
(#1 - #6 = tributary identification,  
P = precipitation, O = outlet, and  
E = evapotranspiration).

# HYDROLOGIC BUDGET

## LAKE OF THE WOODS



INPUT



OUTPUT

TABLE 2.  
Lake of the Woods hydrologic budget.

Source	Volume ( $\text{m}^3 \text{ y}^{-1}$ ) *
Tributary #1	$7.992 \times 10^5$
Tributary #2	$1.667 \times 10^5$
Tributary #3	$7.011 \times 10^5$
Tributary #4	$1.716 \times 10^6$
Tributary #5	$1.387 \times 10^6$
Tributary #6	$1.373 \times 10^5$
Precipitation	$1.790 \times 10^6$
Groundwater and Runoff**	-
Evapotranspiration	$-1.002 \times 10^6$
Outflow	$-5.691 \times 10^6$

\* See text for details of calculations

\*\* Assumed to be zero, since outflow volume was set equal to inflow volume.

## VIII. NUTRIENT BUDGETS

## A. Phosphorus

Construction of the phosphorus budget involved quantification of the phosphorus flux through the lake system using the equation:

$$M_p = Q + P + R + S + DF - O \pm Sd \pm G \quad 1)$$

where

$M_p$  - net loading of phosphorus ( $\text{kg P y}^{-1}$ )

$Q$  = streamflow input ( $\text{kg P y}^{-1}$ )

$P$  = precipitation input ( $\text{kg P y}^{-1}$ )

$R$  = direct runoff input ( $\text{kg P y}^{-1}$ )

$S$  = septic input ( $\text{kg P y}^{-1}$ )

$DF$  = dry fallout input ( $\text{kg P y}^{-1}$ )

$O$  = outflow ( $\text{kg P y}^{-1}$ )

$Sd$  = sediment ( $\text{kg P y}^{-1}$ )

$G$  = groundwater ( $\text{kg P y}^{-1}$ )

Groundwater and direct runoff contributions were considered insignificant since they were assumed to contribute no water loading to the lake. Sediment loading was found from Equation 1 by difference. All other values were measured or calculated directly.

Total phosphorus loading from the tributary streams,  $Q$ , was determined by multiplying average yearly phosphorus concentration ( $\text{mg P m}^{-3}$ ) by total yearly water flow ( $\text{m}^3 \text{y}^{-1}$ ). The resultant yearly stream phosphorus loads ( $\text{kg P y}^{-1}$ ) are shown in Table 3.

Phosphorus input from direct precipitation was estimated using a rainwater phosphorus concentration of  $0.07 \text{ mg l}^{-1}$  (Jorgensen 1980). The total volume of precipitation falling upon the lake surface

TABLE 3.

Yearly phosphorus flux in the streams  
of Lake of the Woods.

Stream	Average Concentration (mg P m <sup>-3</sup> )	Phosphorus flux (kg P y <sup>-1</sup> )
Tributary #1	58.1	46.4
Tributary #2	47.7	8.0
Tributary #3	179.1	125.6
Tributary #4	132.5	227.4
Tributary #5	132.1	183.2
Tributary #6	70.4	9.7
Outlet	53.4	303.9

$(1.790 \times 10^6 \text{ m}^3 \text{ y}^{-1})$  was multiplied by  $0.07 \text{ g m}^{-3}$  to yield a total phosphorus loading of  $125.3 \text{ kg y}^{-1}$ .

Input from dry fallout was calculated based on a phosphorus loading coefficient of  $0.08 \text{ g m}^{-2} \text{ y}^{-1}$  (Rast and Lee 1978). This value was multiplied by the surface area of the lake ( $1.643 \times 10^6 \text{ m}^2$ ) to give an estimated phosphorus loading of  $131.4 \text{ kg y}^{-1}$ .

Septic phosphorus loading was calculated according to the method of Reckhow and Simpson (1980) using the equation:

$$S = E_s \times C_t \times (1 - SR) \quad 2)$$

where

$S$  = total phosphorus input ( $\text{kg y}^{-1}$ )

$E_s$  = export coefficient for septic fields  
 $\{\text{kg}(\text{capita-y})^{-1} \text{ y}^{-1}\}$

$C_t$  = total capita-years

$SR$  = soil retention coefficient (unitless)

$C_t$  is a weighted average of the total number of residents impacting the lake during the year. For Lake of the Woods it was determined to be 1500 (see Appendix V). An export coefficient for septic fields of  $0.3 \text{ kg}(\text{capita-y})^{-1} \text{ y}^{-1}$  and a soil retention coefficient of 0.50 were selected for our estimate (Reckhow and Simpson 1980). Septic loading for Lake of the Woods then becomes:

$$S = (0.3 \text{ kg capita-y}^{-1} \text{ y}^{-1}) \cdot (1500) \cdot (1 - 0.50) = 225.0 \text{ kg y}^{-1} \quad 3)$$

Lake of the Woods had a total phosphorus input from all sources of  $1082.0 \text{ kg y}^{-1}$ . The largest inputs were from septic loading ( $225.0 \text{ kg y}^{-1}$ ) and tributary #4 ( $227.4 \text{ kg y}^{-1}$ ); each of these represents 21% of the total phosphorus entering the system. The next largest input, 17%, was contributed by tributary #5 which transported

183.2 kg y<sup>-1</sup> of phosphorus to the lake. Precipitation, tributary #3 and dry fallout each contributed 12% to the phosphorus budget of the lake. The remaining fraction of phosphorus is divided among tributaries #1, #2, and #6. Figure 9 shows the relative contributions of each source to the phosphorus loading of the lake.

The measured phosphorus loss via the outlet of Lake of the Woods was only 303.9 kg y<sup>-1</sup>. This indicates that 778.1 kg y<sup>-1</sup> or approximately 72% of the phosphorus entering the lake is retained within the system. Calculations indicate that only minor amounts of phosphorus within the lake are channeled into plant and animal biomass. Table 4 shows that a full 90.8% of the phosphorus pool in the lake is tied up in the sediments. Hence, on a yearly basis the sediments serve as a net sink of phosphorus for Lake of the Woods. A complete summary of the phosphorus budget of Lake of the Woods is presented in Table 5.

Mass phosphorus loading is conveniently expressed on an areal basis. Areal phosphorus loading (L) is calculated as:

$$L = \frac{M_p}{A_o} \quad 4)$$

where

L = areal phosphorus loading (g m<sup>-2</sup> y<sup>-1</sup>)

M<sub>p</sub> = mass loading of phosphorus (g y<sup>-1</sup>)

A<sub>o</sub> = lake surface area (m<sup>2</sup>)

For Lake of the Woods this becomes

$$L = \frac{1.082 \times 10^6}{1.643 \times 10^6} = 0.66 \text{ g m}^{-2} \text{ y}^{-1} \quad 5)$$

TABLE 4.  
Compartmentalization of phosphorus  
within Lake of the Woods.

Source	kg P *	%
fish	1099.	6.1
plankton/water	509.	2.8
macrophytes	40	0.2
macrobenthos	15	0.1
sediments	16,393.	90.8

\* See Appendix VI for details on the calculations of these values.

Fig. 9. Phosphorus inputs to Lake of the Woods.  
(#1-#6 = tributary identification, P =  
precipitation, DF = dry fallout, and  
S = septic.)

PHOSPHORUS INPUTS  
LAKE OF THE WOODS

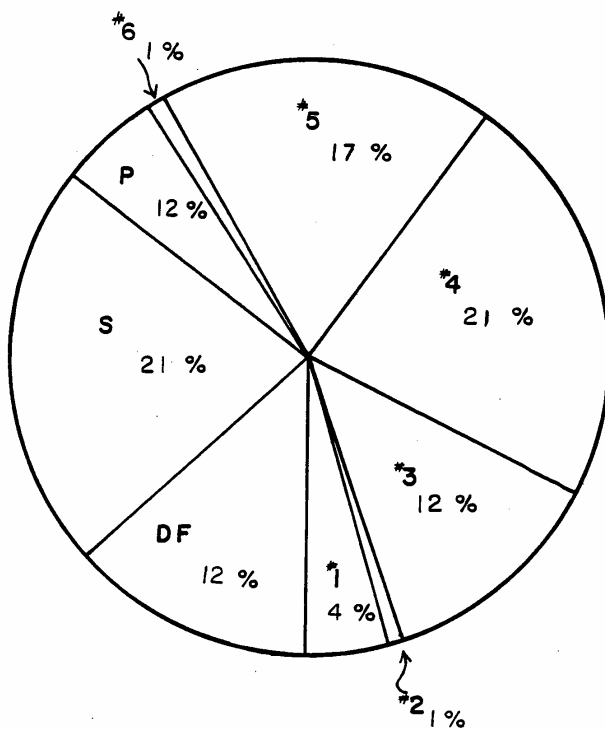


TABLE 5.

Lake of the Woods phosphorus budget.

Source	Loading (kg P y <sup>-1</sup> )
Tributary #1	46.4
Tributary #2	8.0
Tributary #3	125.6
Tributary #4	227.4
Tributary #5	183.2
Tributary #6	9.7
Precipitation	125.3
Dry Fallout	131.4
Septic Loading	225.0
Outlet	-303.9

$$M_p = 1.082 \times 10^3 \text{ kg P y}^{-1}$$

This value is close to the low estimate of  $0.50 \text{ g m}^{-2} \text{ y}^{-1}$  phosphorus loading predicted from the modelling techniques of Reckhow and Simpson (1980) using land-use categories for the watershed (see Appendix V).

The above phosphorus budget considers only external sources of phosphorus loading to Lake of the Woods. Using the model of Reckhow and Simpson (1980), we can estimate the magnitude of internal phosphorus loading from the sediments (see Appendix V). The measured weighted average annual in-lake phosphorus concentration was  $64.9 \text{ } \mu\text{g l}^{-1}$ . From Reckhow and Simpson's (1980) model we know that:

$$.065 = \frac{L}{11.6 + 1.2 (4.07)}$$

where  $L$  = areal phosphorus loading ( $\text{g m}^{-2} \text{ y}^{-1}$ )

Solving for  $L$ , we determined that the total mass loading of phosphorus to Lake of the Woods was  $1.758 \times 10^6 \text{ g y}^{-1}$ . The measured external loading was  $1.012 \times 10^6 \text{ g y}^{-1}$ . Hence, the difference in these values represents the contribution of phosphorus released from the sediments

to the annual mass phosphorus loading. Estimated sediment phosphorus loading was  $7.45 \times 10^5 \text{ g y}^{-1}$ , and represents 42% of the total mass phosphorus loading to Lake of the Woods. This is the single largest phosphorus source, and strongly merits management consideration.

#### B. Nitrogen

Construction of the nitrogen budget involved quantification of the nitrogen flux through the lake system using the equation:

$$M_n = Q + P + S + DF - O \pm Sd \quad (6)$$

where

$M_n$  = net loading of nitrogen ( $\text{kg N y}^{-1}$ )

$Q$  = streamflow N input ( $\text{kg N y}^{-1}$ )

$P$  = precipitation input ( $\text{kg N y}^{-1}$ )

$S$  = septic input ( $\text{kg N y}^{-1}$ )

$DF$  = dry fallout input ( $\text{kg N y}^{-1}$ )

$O$  = outflow ( $\text{kg N y}^{-1}$ )

$Sd$  = sediment ( $\text{kg N y}^{-1}$ )

Groundwater and direct runoff contributions were considered insignificant since they were assumed to contribute no water loading to the lake.

Sediment loading was found from Equation 6 by difference. All other values were measured or calculated directly.

One important aspect of the above nitrogen budget needs to be mentioned. The input of nitrogen to the lake via nitrogen fixation and the loss of nitrogen from the lake via denitrification processes were assumed to be about equal thus cancelling each other in budgetary calculations. Hence, these terms do not appear in Equation 6. This assumption seems valid because: 1) published nitrogen budgets show

these two processes to be fairly equal and 2) the very high loadings of nitrogen from other sources means that both nitrogen fixation and denitrification probably compose only a small fraction of the nitrogen budget.

Inputs and outputs of total nitrogen were determined by summation of the loading of each nitrogen form:

$$TN = NH_3 + NO_2 + NO_3 + ON \quad 7)$$

where

$TN$  = total nitrogen ( $kg\ y^{-1}$ )

$NH_3$  = total ammonia ( $kg\ y^{-1}$ )

$NO_2$  = total nitrite ( $kg\ y^{-1}$ )

$NO_3$  = total nitrate ( $kg\ y^{-1}$ )

$ON$  = organic nitrogen ( $kg\ y^{-1}$ )

Calculations of the nitrogen budget are based upon total nitrogen values. All future references to nitrogen refer to total nitrogen (TN).

Measurements of nitrogen concentrations were made from December, 1980, through August, 1981. The weighted yearly average concentration of the six inlets and Outlet of Lake of the Woods are presented in Table 6. Total nitrogen loading from the tributary streams,  $Q$ , was determined by multiplying average yearly nitrogen concentration ( $mg\ N\ m^{-3}$ ) by total yearly water flow ( $m^3\ y^{-1}$ ). The resultant yearly total nitrogen loads ( $kg\ N\ y^{-1}$ ) are also shown in Table 6.

Total nitrogen input from precipitation was estimated using a rainwater nitrogen concentration of  $1.0\ mg\ l^{-1}$  (Jorgensen 1980). This was multiplied by lake surface precipitation volume ( $1.790 \times 10^6\ m^3\ y^{-1}$ ) to yield a total nitrogen loading of  $179.0\ kg\ y^{-1}$ .

Determination of nitrogen input from dry fallout was made using a nitrogen loading coefficient of  $1.6 \text{ g m}^{-2} \text{ y}^{-1}$  (Rast and Lee 1978). A total estimated nitrogen input of  $2629 \text{ kg y}^{-1}$  was derived by multiplying the loading coefficient by the lake surface area ( $1.643 \times 10^6 \text{ m}^2$ ).

Nitrogen export from septic fields was calculated by a modification of Reckhow and Simpson's (1980) equation in the form:

$$N = E_n \times C_t (1 - SR) \quad 8)$$

where

$N$  = total nitrogen input ( $\text{kg y}^{-1}$ )

$E_n$  = nitrogen export coefficient for septic fields  
 $\{\text{kg}(\text{capita-y})^{-1} \text{ y}^{-1}\}$

$C_t$  = total capita-years

$SR$  = soil retention coefficient (unitless)

Total capita-years ( $C_t$ ), the weighted average number of residents impacting Lake of the Woods during the year, was equal to 1500 as determined previously. An export coefficient ( $E_n$ ) of  $2.2 \text{ kg}(\text{capita-y})^{-1} \text{ y}^{-1}$  (Hook et al. 1978) and a soil retention coefficient of 0.0 were used. The equation for septic nitrogen loading thus becomes:

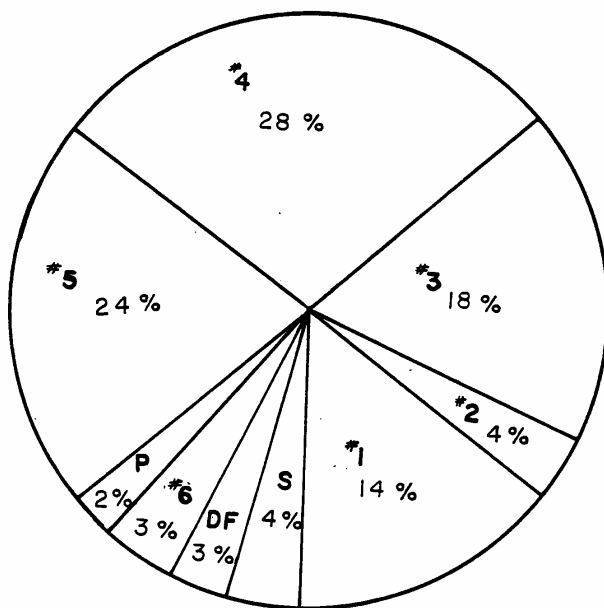
$$N = \{2.2 \text{ kg}(\text{capita-y})^{-1} \text{ y}^{-1}\} \cdot (1500) \cdot (1-0.0) = 3300 \text{ kg y}^{-1} \quad 9)$$

Lake of the Woods had a total nitrogen input of  $8.259 \times 10^4 \text{ kg y}^{-1}$ . The largest input was from tributary #4 which contributed approximately  $2.28 \times 10^4 \text{ kg N y}^{-1}$  or 28% of the loading. Tributary #5 was a close second, contributing 24% or  $1.97 \times 10^4 \text{ kg N y}^{-1}$ . Tributaries #3 and #1 added 18% and 14%, respectively, of the yearly nitrogen flux. Tributary #2, septic loading, dry fallout, precipitation, and tributary #6 each contributed 4% or less to the nitrogen budget. The relative contribution of each source to the nitrogen loading of Lake of the Woods is shown in Figure 10.

Fig. 10. Nitrogen inputs to Lake of the Woods.  
(#1-#6 = tributaries, P = precipitation, DF = dry fallout, and S = septic).

## NITROGEN    INPUTS

## LAKE OF THE WOODS



The outlet transported  $3.89 \times 10^4$  kg of nitrogen out of the lake. Organic nitrogen (74%) and nitrate (25%) were the major nitrogen fractions being lost. Organic nitrogen is exported in a much greater proportion than it is imported. Apparently, inorganic nitrogen forms are converted to organic compounds through biological activity as they pass through the lake system.

Mass nitrogen loading to Lake of the Woods totaled  $8.89 \times 10^4$  kg  $y^{-1}$ . Only 44% of this total is removed through the outflow. The remaining 56% ( $4.98 \times 10^4$  kg  $y^{-1}$ ) is retained in the lake system. A complete summary of the nitrogen budget is presented in Table 7.

Areal nitrogen loading to Lake of the Woods can be calculated from

$$L_n = \frac{M_n}{A_o} \quad (10)$$

where

$L_n$  = areal nitrogen loading ( $g \ m^{-2} \ y^{-1}$ )

$M_n$  = mass loading of nitrogen ( $g \ y^{-1}$ )

$A_o$  = surface area of lake ( $m^2$ )

For Lake of the Woods this becomes:

$$L_n = \frac{8.89 \times 10^7}{1.643 \times 10^6} = 54.1 \ g \ m^{-2} \ y^{-1} \quad (11)$$

This value can be divided by the areal phosphorus loading of  $0.66 \ g \ m^{-2} \ y^{-1}$  (see page 78) to yield an N:P loading ratio of 82:1. This tremendous nitrogen input leaves little doubt that Lake of the Woods is phosphorus limited.

TABLE 6.  
Yearly nitrogen flux in the streams of  
Lake of the Woods.

Stream	Average TN Concentration (mg N m <sup>-3</sup> )	Nitrogen flux (kg N y <sup>-1</sup> )
Tributary #1	$1.48 \times 10^4$	$1.18 \times 10^4$
Tributary #2	$1.75 \times 10^4$	$2.91 \times 10^3$
Tributary #3	$2.17 \times 10^4$	$1.52 \times 10^4$
Tributary #4	$1.33 \times 10^4$	$2.28 \times 10^4$
Tributary #5	$1.42 \times 10^4$	$1.97 \times 10^4$
Tributary #6	$1.78 \times 10^4$	$2.45 \times 10^3$
Outlet	$6.83 \times 10^3$	$3.89 \times 10^4$

TABLE 7.

Lake of the Woods nitrogen budget

Source	Loading (kg N y <sup>-1</sup> )
Tributary #1	$1.81 \times 10^4$
Tributary #2	$2.91 \times 10^3$
Tributary #3	$1.52 \times 10^4$
Tributary #4	$2.28 \times 10^4$
Tributary #5	$1.97 \times 10^4$
Tributary #6	$2.45 \times 10^3$
Precipitation	$1.79 \times 10^3$
Dry fallout	$2.63 \times 10^3$
Septic loading	$3.30 \times 10^3$
Outlet	$3.89 \times 10^4$

$$M_n = 8.89 \times 10^4 \text{ kg N y}^{-1}$$

## IX. MANAGEMENT AND RESTORATION STRATEGIES

The management of Lake of the Woods involves both short term and long term strategies. Short term strategies are designed to provide quick resolution of a problem. However, they are cosmetic in nature, addressing the symptoms, not the cause, of the problem. Hence, they must be repeated on a regular basis. Long term strategies, on the other hand, address the cause of the problem and seek to remedy that cause so that problems do not reoccur. Long term strategies usually take more time to implement and may take years for the results to be seen.

The proposed management recommendations for Lake of the Woods fall into both short term and long term strategies. Before we can consider specific strategies, however, we need to review the relative trophic condition of Lake of the Woods to provide a background and framework for developing these management strategies.

Virtually all of the lakes in Indiana ( $n = 413$ ) were classified according to trophic status and morphometry by Torke and Senft (1979). The trophic condition of each lake was estimated using Bonhomme's Trophic Index. Lakes were assigned eutrophy points based upon nutrient concentrations, dissolved oxygen levels, water transparency, and plankton community composition (Torke and Senft 1979). Possible scores ranged from a minimum of 0 to a maximum of 75. Lake of the Woods was classified using data supplied by the Indiana State Board of Health, and was assigned a score of 42 which placed it into lake category VII B. This category consists of eutrophic lakes which have moderate to severe water quality problems.

Recalculation of eutrophy points for Lake of the Woods based upon data from this study suggest that the condition of the lake has not changed considerably. Further support for this idea comes from a comparison of the chlorophyll a and total phosphorus values for Lake of the Woods with published criteria. According to Wetzel (1975), the value of both these parameters place Lake of the Woods in the eutrophic category. This again suggests that the water quality of Lake of the Woods has not changed in recent years.

In conclusion, our data suggest that Lake of the Woods is currently a eutrophic lake. It has average to below average water quality when compared to most other lakes in Indiana. Management strategies should, therefore, address the major problems that do occur (i.e. short term solutions), and also focus upon improving the water quality of the lake to allow attainment of maximum recreational and aesthetic benefits.

#### Proposed Strategies

Results of this study show that Lake of the Woods is phosphorus limited. This is indicated both by the nitrogen:phosphorus loading ratios as well as the in-lake N and P concentrations. Further evidence comes from the algal assay test which also showed that phosphorus was the limiting nutrient. Hence, long term management strategies should focus upon phosphorus control.

Implementation of short term strategies for Lake of the Woods should focus upon controlling lake macrophytes ("weeds") and reducing algal blooms. Our data document the large macrophyte and algal problems that occur in Lake of the Woods. Short term nutrient control may help alleviate both of these problems.

TABLE 8.

Proposed management strategies for Lake of the Woods.\*

Strategy	Nature	Cost **	Effectiveness
1. Weed barrier	Short to intermediate term	\$0.02/ft <sup>2</sup> /year	100% macrophyte control
2. Chemical	Short term	\$0.006/ft <sup>2</sup> /year	75-100% macrophyte control
3. Harvesting	Short term	Variable	50-85% macrophyte control
4. Chemical precipitation	Short to intermediate term	High	50-80% in-lake phosphorus reduction; retardation of sediment P release
5. No till agriculture	Long term	Unknown	80-90% phosphorus removal on complying areas
6. Buffer zones	Long term	Unknown	25-75% phosphorus removal on complying areas
7. Wetland zones	Long term	Unknown	Up to 95% reduction in stream phosphorus concentration
8. Sewerage diversion	Long term	High	Maximum health protection, reduction in phosphorus input
9. Development restriction	Long term	Unknown	Unknown

\* See text for details of each strategy.

\*\* Labor costs not included.

After lengthy examination of all possible management alternatives, nine management strategies for Lake of the Woods are proposed. These are presented in Table 8. Four of these are short term strategies and five are long term strategies. Detailed discussion of each strategy follows.

#### 1. Weed Barriers

The morphometry of Lake of the Woods and its associated plant community make it ideally suited to the use of vinyl-coated fiberglass mesh screening as a means of aquatic macrophyte (weed) control. Commercially, the product is known as 'Aquascreen'. It is a negatively bouyant, polyvinylchloride coated fiberglass mesh having 64 apertures per  $\text{cm}^2$ , with each opening measuring approximately  $1 \text{ mm}^2$ . The material can be installed early in the spring by anchoring it to the bottom in direct contact with the sediments. It prevents macrophyte growth in two ways: 1) by reducing the available sunlight 50% to 60%, and 2) by forming a physical barrier to plant growth. Aquascreen can be installed later in the season directly over existing plants. Since it is heavier than water, it compresses the plants to the bottom where they die. The gases resulting from decomposition can harmlessly escape into the water through the mesh material. Special permits are not required to use this material.

Independent scientific investigations have shown fiberglass mesh screens to be an extremely successful and ecologically sound method of aquatic macrophyte control. Perkins (1980) states, "...the screens were highly effective for removing nuisance conditions associated with aquatic plant growth, maintaining a plant-free water column for the duration

of placement, and significantly reducing regrowth after panel removal (Perkins, 1980; Boston, 1980)." Placement of the screens in early spring (April) resulted in a 78% to 100% reduction in plant biomass when compared to untreated control plots (Perkins, 1980). Mayer (1978) found that the screens completely eliminated rooted aquatic plants within a three-week time period, and limited regrowth of weeds to about 5% of normal plant cover. Aquascreen is effective in controlling numerous plant species including Myriophyllum spicatum (milfoil), Potamogeton crispus (curlyleaf pondweed), and Ceratophyllum demersum (coontail).

Aquascreen is commercially available in rolls 100 feet long by 7 feet wide. It is manufactured by the Menardi-Southern Corporation of Augusta, Georgia. We have contacted a local distributor (Aquatic Control, Inc., P.O. Box 100, Seymour, Indiana 47274, 812/497-2410) and obtained the following price quotes:

Aquascreen Rolls (7' x 100')	
1 - 9 rolls	\$140.00 each
10 or more rolls	\$125.00 each

The minimum life expectancy of each roll is 10 years. Hence, macrophyte control could be accomplished for  $\$0.02/\text{ft}^2/\text{year}$  ( $\$140.00/\text{roll} \times \text{roll}/700 \text{ ft}^2 \times 1/10 \text{ years}$ ). This is a very economical means of aquatic plant control.

It must be stressed that normal plant communities are essential components of lake ecosystems. Although Lake of the Woods does have a severe macrophyte problem, we do not advocate total plant control. The plant community should be managed to provide for healthy fish populations and good recreational usage of the lake. This makes the limited use of weed barriers especially inviting. One roll could be placed parallel

to a dock and provide a weed-free channel to deeper water without harming the adjacent beneficial weed beds.

## 2. Herbicides

Herbicides have become popular as a means of aquatic macrophyte control. There are several types of herbicides that can be used. Most act to inhibit the photosynthetic process of the growing plants and are thus most effective during active growth of the young macrophytes. If applied properly, herbicides are extremely effective at controlling macrophyte growth.

Although herbicides are extremely effective at controlling aquatic plants, there are several factors which preclude us from recommending them unilaterally. They are expensive to use and provide cosmetic results which are only temporary. Plants killed by these chemicals quickly decompose and release their stored nutrients back into the water, often stimulating further plant growth. When treatments are done on a large scale, the process of decomposition often leads to anoxic conditions which are harmful to fish and other organisms (Nichols and Keeny 1976). Some herbicides are directly toxic to the fish and plankton communities of the lake (Kenaga and Moolenaar 1979), and in some instances the use of two or more herbicides in combination has resulted in a negative synergistic effect. In addition, the ramifications of long term treatment using many currently available herbicides is not fully documented. These chemicals must be used with caution to avoid ecological problems within the lake system.

The herbicide of choice for Lake of the Woods would most likely be 2,4-D (2,4-D butoxyethanol ester) since this chemical has been used with

success in the control of Myriophyllum spicatum in many lakes. When applied in a granular formulation (commercially available as Aqua-Kleen 20), 2,4-D has proven effective in killing most of the roots and shoots of M. spicatum in treated areas (Goddard 1980). This chemical is advantageous in that it can be applied rapidly to large areas without inducing plant fragmentation. However, it cannot be applied in areas where the public may be exposed to the herbicide in detectable levels (Newroth 1980). The cost of this chemical is (prices from Aquatic Control, Inc.):

Aqua-Kleen 20 (50 lb. bag)

1 - 19	\$51.00
20 or more	\$49.00

The suggested application rate is 100 - 200 lbs. per surface acre (1 - 2 lbs. per 430 ft<sup>2</sup>). Using a medium value of 150 lbs. per acre, two treatments annually, and labor supplied by the lake association, the cost of herbicide control with 2,4-D becomes

$$2 \times \frac{\$49.00}{50 \text{ lb.}} \times \frac{150 \text{ lbs.}}{\text{acre}} \times \frac{1 \text{ acre}}{43560 \text{ ft}^2} = 0.006$$

or \$0.006 ft<sup>-2</sup> y<sup>-1</sup>. This is material cost only. Should the association desire to contract a private company to provide the application equipment and labor, the application cost (prices from Aquatic Control, Inc.) in addition to the chemical cost for a single treatment would be:

$$X = 365 + (50 \cdot Z) \quad 1)$$

where

X = application labor cost (in dollars)  
Z = surface area treated (acres)

Thus, the total cost (material and labor) required to treat 10 acres twice a year would be approximately \$4670.00 (or \$0.011 ft<sup>-2</sup> y<sup>-1</sup>).

This price can be expected to increase as chemical and labor costs rise.

Considering the negative potential side effects of chemical herbicide treatment and the high degree of public usage of Lake of the Woods, we recommend using this or any other herbicide only with extreme caution. Professional assistance should be obtained to design a specific herbicide program.

### 3. Harvesting

We suggest that macrophyte harvesting be used as a component of the lake management scheme since it is highly effective at controlling nuisance plant growth. Our data on the plant community dynamics of Lake of the Woods should enable the maximization of the effectiveness of this program.

The Property Owners Association of Lake of the Woods is currently considering the purchase of a mechanical weed harvester. One of the least expensive and most popular commercially available harvesters is the Aqua-Marine Chub. This manual unloading harvester cuts from 1/3 to 1/2 acre per hour, depending upon ambient conditions. Current cost of this device is around \$18,000 (Ron Kerlin, Aquatic Weed Control, Inc., RR 2 Box 414, Syracuse, Indiana 46567, 219/856-2921). The minimum life expectancy of the unit is seven years. In addition to several funding options, Aqua-Marine will consider leasing the machine with an option to buy (contact Ron Kerlin for additional information). This type of harvester has been used very successfully in Center Lake near Warsaw, Indiana. Enquiries directed to the Center Lake Association could provide valuable "hands-on" information about the Chub and its operation (contact John Kleeman, Center Lake

Association, P.O. Box 64, Warsaw, Indiana 46580, 219/267-6320).

An alternative to purchasing a harvester would be to contract a private firm to harvest the weeds in selected areas. One regional firm (Aquatic Weed Control, Inc.) provided the following cost estimates:

<u>harvesting time (h)</u>	<u>cost per hour (\$)</u>
less than 10	85
10	80
more than 10	75

plus a transport charge of \$0.50 cents per mile from Syracuse, Indiana. They harvest about 1/2 acre per hour and deposit the plant material on the shore at selected sites. The local association is responsible for the disposal of the weeds.

Mechanical harvesting is advantageous because it allows immediate removal of the macrophyte biomass and the nutrients contained therein without the possible negative side effects of herbicides (Brooker and Edwards 1975). In addition, it is a site-specific control measure and does not present a hazard to non-target weed beds. There is also a "carry-over effect" that results from weed harvesting programs. Nichols and Cottam (1972) found reduced macrophyte biomass in the year following the implementation of a harvesting program, with the most significant long term effects occurring in deeper weed beds (approximately 1.5 m of water). They determined that three harvests the previous year reduced biomass most effectively in their study lake.

Although we endorse macrophyte harvesting as a short term management strategy in Lake of the Woods, several potential negative attributes of the technique must be considered. First and most importantly, the dominant plant species in the lake is Myriophyllum spicatum (Eurasian milfoil). This species reproduces asexually by fragmentation during

the growing season (Giesy and Tessier 1979). Therefore, the plant fragments generated by the harvesting process which are not collected will float to other suitable areas and continue growing into new mature plants. This is obviously counterproductive to the intent of the program. Another potential problem is the disposal of the plant material once it is removed from the lake. It should be removed from the lakeshore, preferably to a location where the nutrients in the tissue will not return to the lake upon decomposition. Steps should be taken to minimize these potential negative effects, therefore allowing plant harvesting to be an integral part of the management of Lake of the Woods.

The overall goal of the harvesting program is to remove plant biomass from desired areas in a manner which maximizes each harvest (Gerloff and Krombholz 1966). This is accomplished most successfully by harvesting during the period of greatest macrophyte biomass and nutrient standing stock. Peak plant biomass and tissue nutrient pools occurred in early June in Lake of the Woods. This timing would be expected to vary somewhat on a yearly basis, but probably not more than a few weeks under normal conditions. Hence, early June would be the optimum time to initiate a macrophyte harvest.

We emphasize that total macrophyte removal is not desirable in Lake of the Woods. Extensive harvesting in shallow littoral areas could contribute to the erosion of littoral sediments by wave action (Dunst, et al. 1974 as reported by Carpenter and Gasith 1977). This sediment resuspension and movement would increase water turbidity and possibly initiate the release of nutrients from the sediments. In addition, extensive macrophyte removal could cause biological damage to the invertebrate and fish communities. The lake property owners

should select strategic sites within the lake and concentrate their harvesting efforts in these locations.

#### 4. Chemical Precipitation

Many of the water quality problems of Lake of the Woods are directly related to the high concentrations of phosphorus in the water column. A management strategy to reduce the concentration of phosphorus in the lake would help alleviate these water quality problems.

Chemical precipitation is a short to intermediate term management technique that reduces in-lake phosphorus concentrations. Chemicals are applied to the lake which cause phosphorus to precipitate out of the water in a particulate floc. This floc gradually settles to the bottom of the lake where it (and the bound phosphorus) remains unavailable. This creates a seal on the sediment surface which retards the release of phosphorus. Hence, the concentration of phosphorus in the lake water as well as phosphorus release from the sediments is greatly reduced.

Numerous chemicals have been suggested and/or used as flocculating agents. One of the most common is some form of aluminum sulfate (known as "alum"). Alum has been used successfully in numerous lake studies (Eisenrich, et al. 1977; Funk, et al. 1977; Kennedy 1978; Sarville 1976). Although the effectiveness of the treatments are variable, some lakes have experienced over a 90% reduction in water column total phosphorus levels (Cooke, et al. 1978; Kennedy 1978; Dunst, et al. 1974). In addition, sediment sealing by alum significantly interrupts the release of phosphorus from the sediments for upto five years (Cooke and Kennedy 1980) and possibly longer.

As a management strategy for Lake of the Woods, chemical precipitation in the form of an alum treatment would be an effective tool. Since over 42% of the annual phosphorus input into the lake comes from sediment release, an alum treatment would significantly reduce phosphorus loading by interrupting this internal recycling. We would expect that an alum application to Lake of the Woods would reduce the in-lake concentration of phosphorus significantly and also help seal the sediments. However, the cost of an alum treatment is not cheap. Using an alum (17% Al) cost of  $\$0.24 \text{ lb}^{-1}$  (Aquatic Control, Inc.) and a volume of  $2.187 \times 10^6 \text{ m}^3$  for the 0-5 ft. lake stratum, we can project the cost for the following treatment levels:

Treatment	Dosage	Cost
low	2 mg Al $\text{l}^{-1}$	\$ 13,604
medium	10 mg Al $\text{l}^{-1}$	\$ 68,022
high	20 mg Al $\text{l}^{-1}$	\$136,044

This cost does not include labor and equipment for application. The correct dosage level must be ascertained using laboratory jar tests before any applications can be performed.

Although the cost of alum treatment is high, we feel that it represents the most practical method of reducing phosphorus concentrations in the lake. The result of this treatment should be to substantially improve the water quality of the lake by significantly reducing phosphorus concentrations. The longevity of the improvement is partially dependent upon the degree to which external phosphorus loading is reduced. Financial support for an alum treatment may possibly be available through the federal government (EPA Clean Lakes Program) or the state government.

## 5. No Till Agriculture

The watershed of Lake of the Woods is predominately agricultural land. This diffuse source of nutrients collectively supplies almost 60% of the annual external phosphorus loading to the lake via the many ditches which drain surrounding farm fields. Since precipitation and dry fallout phosphorus inputs are impossible to control, attempts to reduce phosphorus inputs into Lake of the Woods should be directed at the dominant source: agricultural fields.

Under most circumstances, runoff water exiting farmed plots transports soil particles and various chemical substances into the streams draining the area. Potential soil loss via erosion in the Lake of the Woods watershed is about 20 tons/acre/year (MACOG 1978). Studies have shown that 90-98% of the phosphorus lost from agricultural fields is bound to the sediment particles (Hubbard, et al. 1982), except during certain winter conditions when upto 33% of the phosphorus may be in dissolved form. Obviously, any reduction in soil loss will bring about concomitant reductions in nutrient export. A management practice focusing on stabilizing topsoil and preventing sediment loss from the fields should benefit both the farmer, by protecting his valuable soil resource, and the lake community, by reducing phosphorus loading to Lake of the Woods.

The single most important land management practice that a farmer can use in no-till plowing. In consideration of local physiographic and soil characteristics, the successful implementation of this tillage method would reduce soil loss 90-95% (personal communication, Steve Boeder, Kosciusko County Soil Conservation Office). Since most of the phosphorus leaving the fields is associated with sediment particles,

a tremendous reduction in phosphorus export would result. The farmer could retain valuable plant nutrients and soil, and the lake benefit by reduced phosphorus loading.

Admittedly, no-till agricultural practices are controversial. Farmers are reluctant to change farming techniques, especially during hard economic times. The costs of such changes are not fully documented. The results of no-till farming, however, far outweigh any negative aspects of its application. Therefore, we strongly urge its adoption by the farmers in the Lake of the Woods watershed. Site specific implementation of this practice should be done in consultation with the Marshall County Soil Conservation Service. They are willing to assist in any possible manner.

#### 6. Buffer Zones

Another long term strategy to reduce phosphorus loading to Lake of the Woods involves the establishment of grassed "buffer zones" along the drainage ditches and their tributary streams. Since 60% of the phosphorus entering the lake from external sources comes from the runoff collected by these streams, any reduction in phosphorus loading to the streams will benefit the lake.

As mentioned above, a large portion of the phosphorus entering the drainage ditches is associated with sediment runoff. A grassed buffer strip 7 to 10 meters wide along each stream bank would significantly reduce field soil loss. A combination of Kentucky 31 Tall Fescue and rye grass is recommended for such buffer zones (personal communication, Steve Boeder, SCS).

This management practice may also be unpopular with local farmers since it takes land out of production. However, the long term benefits

of reduced soil erosion and phosphorus loading to the streams argue for its implementation.

#### 7. Wetlands

Another long term strategy for nutrient reduction is the use of wetlands as a biological filter. It is well documented that the natural vegetation of wetlands serve to remove nutrients from water and "tie them up" in plant tissues. This removal is quite efficient and significant reductions in nutrient loadings to the lake can be accomplished.

It is unfortunate that many of the ditches do not flow through wetlands before entering Lake of the Woods. We strongly urge the establishment and protection of wetlands on drainage streams as a long term management strategy. The cost of this strategy is unknown, but involves only a minimal amount of land. Every effort should be made to establish and protect wetland areas so that they can serve as efficient nutrient traps for the protection of Lake of the Woods.

#### 8. Sewerage Diversion

Our study indicates that the residential septic systems surrounding Lake of the Woods are not functioning properly at this time. Bacteriological surveys produced indications of malfunctioning systems, and make it quite clear that Lake of the Woods has severe public safety problems. Nutrient inputs into the lake from septic systems are also substantial in relation to other sources. Over 22% of the external phosphorus loading to the lake comes from septic systems. Taking all these factors into consideration, we strongly urge that the Lake of the Woods community construct a sewer system to divert septic wastes away from the lake.

The cost of sewerage the lake community and connecting to the Bremen Sewage Treatment Facility would be very high. Previous studies (MACOG 1978) recommended a plan costing \$1,073,800 (1978 dollars). Allowing for the double-digit inflation of the last several years, the projected cost could be assumed to be almost twice the 1978 figure. Despite this fact, sewerage diversion remains an absolute necessity for improving the bacteriological safety of Lake of the Woods as well as reducing external phosphorus loadings.

#### 9. Development Restrictions

Virtually all of the shoreline of Lake of the Woods has been developed for residential usage. This extensive development has caused the lake to be over-burdened with an extensive residential population. We would strongly urge that future development along Lake of the Woods be severely curtailed. The high water table immediately surrounding the lake appears to be saturated with respect to septic fields and further development can only aggravate the situation. In fact, the Marshall County Health Department has on occasion denied permits for septic systems around Lake of the Woods because of seasonally high water tables (MACOG 1978).

#### Additional Considerations

The use of sediment retention basins to reduce siltation and phosphorus loading to Lake of the Woods from surface streams is not a recommended management strategy. Although there is significant sediment and nutrient loading from these streams, calculations indicate that the construction of sediment retention basins would be impractical. The necessary physical dimensions of basins on each major tributary are:

Tributary Number	Basin Surface Area (acres)*	Volume (m <sup>3</sup> )
4	7.1	$8.58 \times 10^4$
5	5.7	$6.93 \times 10^4$
1	3.3	$3.99 \times 10^4$
3	2.9	$3.50 \times 10^4$

\* all basins 3 m (10 ft) deep

The costs and logistic problems involved with excavating such huge quantities of soil are obvious. Furthermore, these structures must be redredged to original depth when their volume is reduced to 50% by accumulated sediment (Novotny and Chesters 1981). Sediment retention basins also selectively retain larger sediment particles and are inefficient in capturing the small flocculant particles that often comprise a significant portion of the stream sediment load. These drawbacks, in addition to other complicating factors, make it evident that sediment retention basins are impractical for use at Lake of the Woods. As previously discussed, the most logical approach involves preventing excessive sediment and nutrient export from agricultural lands in the first place (through no-till agriculture and grassed buffer zones). This approach is much better than attempting to cosmetically treat the situation with sediment retention basins.

#### Predicted Lake Response

The nine management strategies listed above have been suggested for implementation on Lake of the Woods. The obvious question and

the one asked most frequently by the concerned citizen is: "Will these strategies work and how much of an improvement in the lake can be expected?"

This question must be answered separately for short term versus long term strategies. In the case of the three short term strategies (weed barriers, herbicides, and harvesting) the answer is quite clear. All three strategies will provide a very significant improvement in the lake (i.e. effective weed control). It must be remembered that these are only cosmetic solutions that must be repeated on a regular basis. For their designated purpose of reducing yearly weed problems, however, they will be effective.

The effects of the intermediate to long term strategies on Lake of the Woods are a little more difficult to analyze. It is a fact that any reduction in phosphorus loading to the lake will be beneficial. The effectiveness of each strategy depends upon the degree of phosphorus removal accomplished by that strategy.

One way to look at the postulated improvements in the water quality of Lake of the Woods is to use a model developed for such purposes (Reckhow and Simpson 1980). Empirically, average annual in-lake phosphorus concentrations can be predicted from the equation:

$$P = \frac{L}{11.6 + 1.2 (q_s)} \quad 2)$$

where

$P$  = mean annual in-lake  $P$  concentration ( $\text{mg l}^{-1}$ )

$L$  = areal  $P$  loading ( $\text{g P m}^{-2} \text{ y}^{-1}$ )

$q_s$  = areal water loading ( $4.07 \text{ m y}^{-1}$ )

Hence, we can look at the change in average in-lake phosphorus concentration resulting from various reductions in phosphorus removal.

The effect of reduced phosphorus loading on the phytoplankton community of Lake of the Woods can also be predicted, although with much less confidence. Empirical relationships have been developed which relate average summer chlorophyll a concentrations (an indicator of algal biomass) to average summer total phosphorus levels in the epilimnion of a lake. One commonly used equation developed by Jones and Bachmann (1976) is:

$$\log \text{Chl. } \underline{a} = -1.09 + 1.46 (\log \text{TP}) \quad 3)$$

where

Chl. a = chlorophyll a concentration ( $\mu\text{g } \ell^{-1}$ )

TP = total phosphorus concentration ( $\mu\text{g } \ell^{-1}$ ),

We will use this relationship cautiously in our discussion, for many lakes respond in unique manners which are not adequately described by this or other current models (Smith and Shapiro 1981).

One other parameter that will be affected by reduced phosphorus loading is Secchi disk transparency. This parameter is meaningful to lake residents, for they can visually see improvements in water clarity. Carlson (1977) empirically related Secchi depth to total phosphorus using the equation:

$$\ln \text{SD} = 3.876 - 0.98 (\ln \text{TP}) \quad 4)$$

where

SD = Secchi depth (m)

TP = total phosphorus concentration ( $\mu\text{g } \ell^{-1}$ )

This can be manipulated to produce the simple predictive equation:

$$SD = 48 \left( 1/TP \right)$$

5)

We can use this relationship to model changes in Secchi depth as a function of total phosphorus.

The effects of various phosphorus reduction strategies on in-lake phosphorus concentration, chlorophyll a concentrations, and Secchi disk transparency are shown in Table 9. Assuming the phosphorus values predicted from equation 2 (Table 9) to be representative of mean epilimnetic summer values, the following predictions can be made. If a 100% reduction in phosphorus input from septic systems was attained, the resultant in-lake phosphorus concentration would be  $57 \mu\text{g l}^{-1}$ . This would yield an average summer Secchi depth of only 0.84 m, just a slight improvement over the measured 1981 value. It would appear, therefore, that a 100% reduction in septic loading alone would result in little qualitative improvement of water transparency. Conversely, a 100% reduction of sediment phosphorus loading to the lake would result in a phosphorus concentration of  $38 \mu\text{g l}^{-1}$  and a Secchi depth of 1.26 m. This represents a significant improvement in the water quality of Lake of the Woods.

It is very difficult to predict the response of macrophyte communities to a reduction in phosphorus input, for many interrelating factors determine their growth response. It is possible that the reduced availability of in-lake phosphorus could reduce macrophyte biomass in certain littoral areas of Lake of the Woods.

TABLE 9.

Predicted response of Lake of the Woods to various hypothetical reductions in phosphorus loading.

Septic seepage (% reduced)	Strategy		Loading (g P m <sup>-2</sup> y <sup>-1</sup> )	In-lake P (µg l <sup>-1</sup> )	Chlor a (µg l <sup>-1</sup> )	Secchi (m)
	Sediment loading (% reduced)	Stream flow loading (% reduced)				
0	0	0	1.07	65	36.05	0.74
100	0	0	0.94	57	29.76	0.84
0	50	0	0.84	51	25.30	0.94
0	100	0	0.62	38	16.46	1.26
100	100	0	0.48	29	11.09	1.66
100	0	50	0.75	45	21.07	1.07
100	0	100	0.57	35	14.60	1.37
100	100	100	0.11	7	1.39	6.86

We feel that significant changes in the water quality of Lake of the Woods would require the installation of a sewerage system (for bacteriological and nutrient concerns) and at least a 50% reduction in sediment phosphorus loading. This would lower the mean annual phosphorus concentration by one-third to one-half and place Lake of the Woods much closer to Vollenweider's (1968) permissible loading range. Two basic questions remain. What is the realistic probability that the degree of abatement necessary to elicit obvious improvements in the water quality of Lake of the Woods can be obtained from these strategies? Furthermore, if these abatement levels cannot be reached, is it still important to make every attempt to reduce the amount of phosphorus exported from the agricultural lands in the watershed?

The answer to the first question is unclear. Certainly, a sewer system would alleviate bacteriological problems in the lake and also reduce phosphorus loading. Alum treatment would be effective at reducing in-lake phosphorus concentrations and retarding the release of phosphorus from the sediments. The effectiveness of this treatment is highly variable, but we could expect to retard phosphorus release from sediments for up to five years.

The reduction of phosphorus loading from the tributary streams by no-till agriculture and grassed buffer zones is also a goal worth striving for. Remember that no-till agriculture and grassed waterways could reduce soil loss by over 90%, and that 90% to 98% of the phosphorus leaving farm fields is bound to these sediment particles. Hypothetically, it is possible to achieve a high level of reduction in the watershed phosphorus inputs into Lake of the Woods with these two management strategies. Realistically, however,

such a high level of abatement is probably not likely due to non-compliance by many farmers.

If these high abatement levels cannot be attained, is it still important to make every attempt to reduce phosphorus and soil export from agricultural lands? The answer is a strong yes! Smaller reductions might not elicit a visual improvement in the water quality of Lake of the Woods, but they definitely will help prevent further degradation of the lake system. Since Lake of the Woods is a net sink for phosphorus inputs, any reduction in phosphorus loading would help to slow the natural eutrophication process. Lake of the Woods currently has poor water quality, and in order to improve that water quality every effort should be made to reduce phosphorus loading by utilizing the long term management strategies proposed in this report.

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APPENDIX I  
METHODOLOGY

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Procedure 1. Program for calculating stream flow.

```

DIMENSION X(25),DEPTH(25),FLOW(25),LOC(6),LDATE(2)
WRITE(5,91)
READ(5,92) (LOC(I),I=1,6)
WRITE(5,93)
READ(5,94) (LDATE(I),I=1,2)
WRITE(5,95)
READ(5,96) STAFF
WRITE(5,101)
READ(5,102) WIDTH
K=INT(WIDTH/0.25)
X(1)=0.0
DEPTH(1)=0.0
FLOW(1)=0.0
X(K)=0.0
DEPTH(K)=0.0
FLOW(K)=0.0
TFLOW=0.0
DO 10 J=2,K
  X(J)=X(J-1)+0.25
  WRITE(5,103) X(J)
  WRITE(5,104)
  READ(5,105) DEPTH(J)
  DEPTH(J)=DEPTH(J)/100.
  WRITE(5,106)
  READ(5,107) FLOW(J)
  FLOW(J)=FLOW(J)*0.3048
  FFLOW=FLOW(J)*DEPTH(J)*0.25
  TFLOW=TFLOW+FFLOW
10 CONTINUE
C
C **** OUTPUT RESULTS ****
C
  PAUSE 'ADJUST PAPER TO TOP OF NEXT PAGE'
  WRITE(5,99)
  WRITE(5,97) (LOC(I),I=1,6), (LDATE(I),I=1,2)
  WRITE(5,98) STAFF, TFLOW
  WRITE(5,99)
  WRITE(5,99)
  WRITE(5,99)
  WRITE(5,99)
  WRITE(5,99)
91 FORMAT(1H+,2X,'LAKE NAME:? ',,$)
92 FORMAT(6A4)
93 FORMAT(1H+,2X,'DATE:? ',,$)
94 FORMAT(2A4)
95 FORMAT(1H+,3X,'STAFF GAUGE READING:? ',,$)
96 FORMAT(F5.1)
97 FORMAT(25X,'STREAM FLOW'/26X,6A4/28X,2A4//)
98 FORMAT(4X,'STAFF GAUGE READING: ',F5.1,5X,'STREAM
  1FLOW (M3/SEC): ',1PE8.2)
99 FORMAT(1X,////)
101 FORMAT(4X,'STREAM WIDTH (IN M ) IS:? ')
102 FORMAT(F5.2)
103 FORMAT(2X,'WIDTH',F5.2,' M')
104 FORMAT(4X,'DEPTH (IN CM) ;? ',,$)
105 FORMAT(F5.1)
106 FORMAT(1H+,4X,'FLOW (IN FT./SEC):? ',,$)
107 FORMAT(F5.2)
  STOP
  END

```

## Procedure 2a

## PHOSPHORUS REAGENTS

REAGENTS:

1. Sulfuric acid, 5N  
bring 140 mls. conc.  $\text{H}_2\text{SO}_4$  to 1 liter with distilled  $\text{H}_2\text{O}$ .  
  
Sulfuric acid, 10.8N (for digesting)  
bring 300 mls. conc.  $\text{H}_2\text{SO}_4$  to 1 liter with distilled  $\text{H}_2\text{O}$ .
- \*ALWAYS ADD ACID TO WATER
2. Antimony potassium tartrate  
dissolve 4.39 gm. of  $\text{K}(\text{SbO})\text{C}_4\text{H}_4\text{O}_6 \cdot 1/2 \text{H}_2\text{O}$  in 200 ml. distilled  $\text{H}_2\text{O}$ . — Store in a dark bottle at 4° C.
3. Ammonium molybdate  
dissolve 20 gm. of  $(\text{NH}_4)_6\text{Mo}_7\text{O}_{24} \cdot 4 \text{H}_2\text{O}$  in 500 ml. distilled water. — Store in a plastic bottle at 4 C.
4. Ascorbic acid, 0.1M  
dissolve 1.76 gm. of ascorbic acid in 100 ml. distilled  $\text{H}_2\text{O}$ . — Stable for about one week at 4 C.
5. Hydrochloric acid, 1N  
bring 84 ml. conc.  $\text{HCl}$  to 1 liter with distilled  $\text{H}_2\text{O}$ .
6. Sodium hydroxide, 10N  
dissolve 400 gm.  $\text{NaOH}$  into 1 liter of distilled  $\text{H}_2\text{O}$ .
7. Phenolphthalein indicator  
dissolve 5 gm. of phenolphthalein in 500 ml. 95% ethyl or isopropyl alcohol and add 500 ml. distilled  $\text{H}_2\text{O}$ .
8. Potassium persulfate  
dissolve 5 gm.  $\text{K}_2\text{S}_2\text{O}_8$  in 100 ml distilled  $\text{H}_2\text{O}$ .
9. Phosphorus solution, 0.25 umoles/ml  
dissolve 33.99 mg.  $\text{KH}_2\text{PO}_4$  in 1 liter of distilled  $\text{H}_2\text{O}$ .
10. Phosphate color reagent  
1-50 ml. 5N  $\text{H}_2\text{SO}_4$   
2-5 ml. Antimony potassium tartrate  
3-15 ml. Ammonium molybdate  
4-30 ml. Ascorbic acid (good for only 7 days)

mix the above reagents carefully. The color reagent is stable for 1 week when stored at 4 C. Should the color reagent become cloudy; heat gently till transparent.

## Procedure 2b

TOTAL  $\text{PO}_4$ 

1. Add 50 ml. of lake water to a 125 erhlermeyer flask.  
(Run standard at the same time; see note #)
2. Add 7.5 ml persulfate solution and 0.5 ml.  $\text{H}_2\text{SO}_4$ . 10.8N.
3. Cover flask with a 50 ml. beaker and autoclave for 1 hour.
4. After samples have cooled, add 2 drops phenolphthalein indicator.
5. Add 10N NaOH dropwise until indicator turns pink.
6. Add 1N HCl dropwise until pink color just disappears.
7. Add 5.0 ml. of phosphate color reagent and allow at least 10 minutes for color development (color is stable for up to 1 hour).
8. Read at 880 mμ against blank \*, and record.

#Standards (0.5, 0.25, 0.15, and 0.10 umoles P) are prepared by adding 2.0, 1.0, 0.6, and 0.4 mls. of a 0.25 umoles P/ml stock solution to 4 flasks. Each flask is then brought to 50 mls. by adding deionized  $\text{H}_2\text{O}$ . Standards are analyzed in exactly the same manner as the unknowns.

\*A blank is prepared exactly as a sample except that no phosphate reagent (step #7) is added. — A turbidity blank (TBL) may be run against a deionized  $\text{H}_2\text{O}$  blank; then the deion. blank should be run against unknowns. Then turbidity value can be subtracted. — Standards may also read against deion. blank.

## Procedure 2c

## SOLUBLE REACTIVE PHOSPHORUS

1. Add 50 mls. of filtered lake<sup>U</sup> water to a 125 erlenmeyer flask.\*
2. Add 5m. of phosphate reagent and mix at once. Allow 10 minutes for color development.
3. Read at 880 against blank.\*\*

\*Standards (0.25, 0.125, 0.063 and 0.05 umoles P) are prepared by adding 2.0, 1.0, 0.6, and 0.4 ml. of 0.125 umole/ml. stock solution to four flasks and bringing volume to 50 ml. with deionized water. Standards are analyzed exactly as unknowns.

\*\*A blank is prepared exactly as a sample except that no phosphate reagent is added.

U Water should be filtered through washed filters (i.e.-filters that have been thoroughly rinsed with  $\text{PO}_4$  free water).

## Procedure 3

## AMMONIA

REAGENTS:

## A. Solution A

Dissolve 10.0 g of phenol (carbolic acid) and 0.05 g of sodium nitroferricyanide in deionized water and bring to 1 l volume.

## B. Solution B

Dissolve 5.0 g of sodium hydroxide and 0.42 of sodium hypochlorite (= 8.4 ml. of bleach) in deionized water and bring to 1 l volume.

## C. Ammonia Solution

Stock solution 1 umole  $\text{NH}_3$ /ml

Dissolve 62.27 mg of  $(\text{NH}_4)_2\text{SO}_4$  in deionized water and bring to exactly 1 l.

Standard solution 0.10 umoles  $\text{NH}_3$ /ml

Add 100 ml. of the Stock ammonia solution to 1 l flask and bring to volume with deionized water.

PROCEDURE:

1. Add 50 ml. of unfiltered water sample to a 125 erlenmeyer flask. #
2. Raise pH with 3N NaOH to between 5 and 7.
3. Add 5.0 ml. of Solution I and 5.0 ml. of Solution II.
4. Cover with a 50 ml. beaker and let stand at room temperature for 4 hours for color development (color is stable for 24 hours).
5. Read at 625 mμ against blank \* and record.

# Standards (0.50, 0.25, 0.15, and 0.10 umoles  $\text{NH}_3$ ) are prepared by adding 5.0, 2.5 1.5, and 1.0 ml. of a 0.10 umole  $\text{NH}_3$ /ml. standard solution to each of four volumetric flasks. Each flask is then brought to 50 ml. by adding deionized water. Standards are analyzed exactly as unknowns.

\* A blank is prepared exactly as a sample except that 50 ml. of deionized water is used instead of 50 ml. of water sample.

REFERENCES:

Chaney, A. L. and E. P. Marboch. 1962. Modified reagents for determination of urea and ammonia. Clinical Chemistry 8(2):130-132.

## Procedure 4

## NITRITE

REAGENTS:

1. Ammonium hydroxide, concentrated.
2. Ammonium chloride - EDTA solution.  
dissolve 13 g ammonium chloride and 1.7 g of disodium ethylenediamine tetracetate in 900 ml deionized water. Adjust the pH to 8.5 with concentrated ammonium hydroxide and dilute to 1 liter with deionized water.
3. Color reagent.  
dissolve 10 g sulfanilamide and 1 g N (1-naphthyl)-ethylene-diamine dihydrochloride in a mixture of 100 ml concentrated phosphoric acid and 800 ml of deionized water and dilute to 1 liter with deionized water.
4. Stock Nitrite solution 1.0 ml = 1.00 mg  $\text{NO}_2\text{-N}$ .  
dissolve 6.072 g  $\text{KNO}_2$  in 500 ml deionized water and dilute to 1 liter. Preserve with 2 ml of chloroform and refrigerate. Stable for approximately 3 months.
5. Standard Nitrite solution 1.0 ml = 0.01 mg  $\text{NO}_2\text{-N}$ .  
dilute 10.0 ml of stock nitrite solution to 1 liter using deionized water.

PROCEDURES:

1. Add 25.0 ml water sample to 75.0 ml ammonium chloride-EDTA solution in 125 ml erlenmeyer flask.\*
2. Mix well and add 4.0 ml of color reagent to sample.
3. Allow 10 minutes for color development and read spectrophotometrically against blank.# at 540 nm. Color is stable for 2 hours.

\* Standards (1.786, 0.893, 0.357, 0.179, and 0.089 umoles  $\text{NO}_2$ ) are prepared by adding 10.0, 5.0, 2.0, and 1.0 and 0.5 ml standard solution to 50 ml deionized water and bringing to volume in 100 ml volumetric flasks. A 25 ml aliquot is then removed and analyzed exactly as unknowns.

# A blank is prepared exactly as sample except that 25.0 ml of deionized water is used instead of water sample.

REFERENCES:

Environmental Protection Agency. 1979. Methods for chemical analysis of water and wastes.

## NITRATE

REAGENTS:

1. Granulated cadmium: 40-60 mesh (E. M. Laboratories, Inc., 500 Exec. Blvd., Elmsford, NY 10523, Cat. 2001 Cadmium, Coarse Powder).
2. Copper-Cadmium: The cadmium granules (new or used) are cleaned with dilute HCl and copperized with 2% solution of copper sulfate in the following manner:
  - A. Wash the cadmium with dilute HCl and rinse with deionized water. The color of the cadmium should be silver.
  - B. Swirl 25 g cadmium in 100 ml portions of a 2% solution of copper sulfate for 5 minutes or until blue color partially fades, decant and repeat with fresh copper sulfate until a brown colloidal precipitate forms.
  - C. Wash the copper-cadmium with deionized water (at least 10 times) to remove all the precipitated copper. The color of the cadmium so treated should be black.
3. Preparation of reduction column.
  - A. Insert glass wool plug into bottom of 50 ml buret (1 cm I.D.) Fill with deionized water. Add sufficient copper-cadmium granules to produce column 15 cm in length. Maintain liquid level above granules to avoid entrapment of air. Wash the column with 200 ml of dilute ammonium chloride-EDTA solution. Place 100 ml glass funnel atop column to assist in the addition of the sample to the buret. Activate the column by passing 100 ml of a solution composed of 25 ml of 1.786 umole  $\text{NO}_3$  standard and 75 ml of ammonium chloride-EDTA solution. Use flow rate of 7 to 10 ml per minute.
4. Ammonium hydroxide, conc.
5. Dilute hydrochloric acid, 6N: Dilute 50 ml of conc. HCl to 100 ml with deionized water.
6. Copper sulfate solution, 2%: Dissolve 20 g of  $\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$  in 500 ml of deionized water and dilute to 1 liter.
7. Ammonium chloride-EDTA solution: Dissolve 13 g ammonium chloride and 1.7 g disodium ethylenediamine tetracetate in 900 ml of deionized water. Adjust the pH to 8.5 with conc. ammonium hydroxide and dilute to 1 liter.
8. Dilute ammonium chloride-EDTA solution: Dilute 300 ml of ammonium chloride-EDTA solution to 500 ml with deionized water.

## Procedure 5 (con't.)

9. Color reagent: Dissolve 10 g sulfanilamide and 1 g n(1-naphthyl)-ethylene-diamine dihydrochloride in a mixture of 100 ml conc. phosphoric acid and 800 ml of deionized water and dilute to 1 liter with deionized water.
10. Stock nitrate solution: Dissolve 7218 g  $\text{KNO}_3$  in deionized water and dilute to 1000 ml. Preserve with 2 ml of chloroform per liter. This solution is stable for at least 6 months. 1.0 ml = 1.00 mg  $\text{NO}_3\text{-N}$ .
11. Standard nitrate solution: Dilute 10.0 ml of nitrate stock solution to 1000 ml with deionized water. 1.0 ml = 0.01 mg  $\text{NO}_3\text{-N}$ .

PROCEDURES:

1. Add 25.0 ml water sample to 75.0 ml ammonium chloride-EDTA solution in 125 erhlermeyer flask.\*
  2. Pour sample into reduction column and collect at rate of 7 to 10 ml per minute.
  3. Discard the first 40 ml and collect 25.0 ml in a volumetric flask.
  4. Pour reduced sample into flask and add 1.0 ml of color reagent. Reduced samples should not stand longer than 15 minutes before the addition of color reagent.
  5. After 10 minutes but before 2 hours, read the absorbance spectrophotometrically at 540 nm against a blank # and record.
- \* Standards (1.786, 0.893, 0.357, 0.179, and 0.089 umoles  $\text{NO}_3$ ) are prepared by adding 10.0, 5.0, 2.0, 1.0 and 0.5 ml of standard  $\text{NO}_3$  solution to 50 ml deionized water and diluting to 100 ml in volumetric flask. A 25.0 ml aliquot is added to 75.0 of ammonium chloride-EDTA and is analyzed exactly as an unknown.
- # A blank is prepared exactly as a sample except that 25.0 ml of deionized water is used.

NOTES ON THE USE OF THE REDUCTION COLUMN:

1. Between samples, run 100 ml of deionized water through column before adding next sample.
2. After all samples are reduced, wash the column with deionized water and fill the buret with dilute ammonium chloride-EDTA solution for storage. Cover with parafilm to prevent evaporation.
3. A well trained technician can operate three columns at a time.

4. The reduction response of each column should be determined initially by running a complete standard curve through each column and comparing the resulting slopes for significant differences. All columns should yield the same response.
5. Periodically recharge the cadmium and repack the columns. Recheck the responses of each column at this time.
6. When processing standards during normal operation, it is best to randomly assign each standard to a different column.
7. Do not allow air to become trapped in the cadmium granules.

REFERENCES:

Environmental Protection Agency. 1979. Methods for the chemical analysis of water and wastes.

## Procedure 6

## TOTAL ORGANIC NITROGEN

REAGENTS:

1. Phenol-alcohol solution  
dissolve 10.0 g of reagent grade phenol in 100 ml. of 95% v/v ethyl alcohol USP.
2. Sodium nitroprusside 0.5%  
dissolve 1.0 g of sodium nitroprusside (nitroferricyanide) in 200 ml. water. Store in amber bottle for maximum of 30 days.
3. Alkaline solution  
dissolve 100.0 g of trisodium citrate and 5.0 g sodium hydroxide in 500 ml water.
4. Sodium hypochlorite  
use commercial hypochlorite (e.g. Chlorox) which should be at least 1.5 N (decomposes).
5. Oxidizing solution  
mix 100.0 ml alkaline solution with 25.0 ml of sodium hypochlorite solution. Use the same day.
6. Nitrogen stock solution 1 mM/ml  
dissolve 66.06 g ammonium sulfate in 900 ml organic-free deionized water and bring to 1 l.
7. Nitrogen standard solution 2.5 uM/ml  
dissolve 2.5 ml of N stock solution in 900 ml organic-free deionized water and bring to 1 l.

PROCEDURES:

1. \*Place 25 ml of lake water into a 125 ml Erlenmeyer flask.
2. Add 1.0 g potassium persulfate ( $K_2S_2O_8$ ).
3. Cover with 50 ml beaker and autoclave at  $121^\circ C$  and 15 psi for 1 hr.
4. Remove and cool to room temperature.
5. Add 0.5 g DeVarda's Alloy to each flask and allow them to sit for 24 hrs. This step is important.
6. Swirl flask gently and remove 1.0 ml aliquot with automatic pipett.

7. Dilute to 25.0 ml volume using deionized water and pour into clean 50 ml beaker.
  8. Add 3 drops of 3N NaOH to sample. Check pH with meter. Add 1 N NaOH dropwise until pH is approximately 5.0 (the addition of 1 N NaOH may not be necessary).
  9. Add 1.0 ml. of phenol-alcohol solution, 1.0 ml of sodium nitroprusside solution, and 2.5 ml of oxidizing solution to each sample in rapid succession.
  10. Read spectrophotometrically after 1 h at 640 nm.
- \* Standard preparation: Prepare standard curve in triplicate of 7.5, 5.0, 2.5, and 1.25  $\mu\text{M}$  by bringing 3.0, 2.0, 1.0 and 0.5 ml of 2.5  $\mu\text{M}/\text{m}/(\text{NH}_4)_2\text{SO}_4$  standard solution to 25 ml in volumetric flask with N-free Insta-pure (Baker) water. Standards are analyzed exactly as unknowns.

NOTE: The value obtained from the standard curve must be corrected for trace amounts of  $\text{NO}_2$ ,  $\text{NO}_3$ , and  $\text{NH}_3$  with the following equation:

$$\text{Total Organic N} = A - (.04 \times ((\text{NO}_2) + (\text{NO}_3) + (\text{NH}_3)))$$

Where A = value from standard curve  
(all concentration values in  $\mu\text{M}/\text{l}$ )

#### REFERENCES:

- Raveh, A. and Y. Avnimelech. 1979. Total nitrogen analysis in water, soil and plant material with persulfate oxidation. *Water Res.* 13:911-912.
- Solorzano, L. 1969. Determination of ammonia in natural water by the phenol hypochlorite method. *Limnol. Oceano.* 14:799.

## Procedure 7

## REACTIVE SILICA

REAGENTS:

## A. Solution A -- 6N HCl

to approximately 400 ml. of deionized water, carefully add 500 ml. of concentrated HCl. Cool, and bring volume to exactly 1000 ml. Store in plastic container.

## B. Solution B -- Ammonium molybdate

dissolve 50.0 g  $(\text{NH}_4)_6\text{Mo}_7\text{O}_{24} \cdot 4\text{H}_2\text{O}$  in approximately 400 ml distilled water. Bring to exactly 500 ml. and store in a plastic container.

## C. Solution C -- Oxilic acid

dissolve 50.0 g  $\text{H}_2\text{C}_2\text{O}_4 \cdot 2\text{H}_2\text{O}$  in approximately 400 ml. distilled water. Bring to exactly 500 ml. and store in a plastic container.

## D. Silica solutions

Stock solution -- 20 umoles/ml

dissolve 5.68 g  $\text{Na}_2\text{SiO}_3 \cdot 9\text{H}_2\text{O}$  in recently boiled and cooled distilled water. Bring volume to exactly 1000 ml. and store tightly capped in a plastic container.

Standard solution - 1 umole/ml

dilute 5.0 ml stock silica solution to 100.0 ml. with freshly boiled and cooled distilled water.

PROCEDURE:

1. To 50.0 ml of filtered water #4 add in rapid succession
  - a) 1.0 ml. solution A
  - b) 2.0 ml. solution B
2. Mix well and allow the solution to stand for 5 to 10 minutes.
3. Add 1.5 ml. oxilic acid solution and mix well.
4. Read the color after 2 minutes but before 15 minutes at 410 nm against a distilled water blank.\*

# IMPORTANT: Filter through millipore, not glass fiber, filters!!!

ψ Standards (0.4, 0.6, 1.0, 2.0 umole  $\text{SiO}_2$ ) are prepared by adding 0.4, 0.6, 1.0, and 2.0 ml of the standard silica solution to each of four flasks. Each flask is then brought to 50 ml by addition of deionized water. Standards are analyzed exactly as unknowns.

\* Prepare a reagent blank by adding 50 ml. of deionized water to a flask and analyzing. This gives background (reagent) contamination. These O.D. values are subtracted from the standard curve O.D. values only. Prepare a turbidity blank by adding 50 ml. filtered sample to a flask, and omitting color reagents.

NOTE:

All flasks graduate cylinders, etc. must not be glass. Polycarbonate or other plastic labware should be used.

## RESIDUE

APPARATUS:

1. 100 ml porcelain crucibles.
2. Drying oven.
3. Muffle furnace.

PROCEDURE:

1. Clean, ash, and pre-weigh crucible.
2. Add 150 ml water sample (2 aliquots of 75 ml) to crucible and evaporate to dryness at 98° C in drying oven for 24 hours.
3. Remove, cool in dessicator, and obtain dry weight.
4. Place into muffle furnace and ash at 550° C for 1 hour.
5. Remove, cool in dessicator, and record ash weight.

CALCULATIONS:

$$\text{Total Residue (mg/l)} = \frac{(A - B) \times 1000}{C}$$

Where A = dry weight of sample + dish (mg)

B = dry weight of dish (mg)

C = volume of sample (ml)

A similar calculation is performed using ash weight.

## CONCENTRATIONS OF PARTICULATE MATERIAL IN LAKE WATER

The concentrations of particulate material larger than  $1\mu$  diameter may be estimated from the weight of the material retained by glass fiber filters.

PROCEDURES:

Preheat on electric muffle furnace until the temperature is stable at  $500^{\circ}\text{C}$ . Place a required number of glass fiber filters in individual folded squares of aluminum foil, number the foil squares, <sup>1</sup> and put them in the oven for 1 hr. to ignite any residual organic material on the filters. Remove the folded foil squares and their filters from the oven and cool to room temperature in a dessicator. Weight each filter to the nearest 0.1 mg. (The filters will be brittle after heating and they should be handled with flat-bladed forceps, so as not to contaminate them.)

To filter a water sample, place a filter on a filter base fitted to a vacuum filtration flask. Moisten the filter with a few ml. of water from a wash bottle, attach the filtration funnel, and filter a measured volume of water. Dismantle the filtration assembly, remove the filter with a forceps, fold it in half, and put it in its numbered foil square. Put the folded foil square with its filter in a drying oven for 1 hr. at  $101^{\circ}\text{C}$ .<sup>3</sup> Cool to room temperature in a dessicator and weigh the filter to the nearest 0.1 mg.

Put the filter back into its folded foil square and heat it in the muffle furnace of 1 hr. at  $500^{\circ}\text{C}$ . to ignite the organic material. Remove from the oven, cool, and weight.<sup>4</sup>

CALCULATIONS:

Calculate the concentrations of particulate material using the following equations.

1. particulate material/liter ( $W_p$ ):

$$W_p = \frac{W_{101} - W_f}{V_f}$$

2. particulate organic material/liter ( $W_o$ ):

$$W_o = \frac{W_{101} - W_{500}}{V_f}$$

3. particulate inorganic material (ash/liter ( $W_i$ ):

$$W_i = W_p - W_o$$

Where  $W_f$  = weight of the filter

$W_{101}$  = weight of the filter after filtration and heating of  $60^{\circ}\text{C}$

$W_{500}$  = weight of the filter after heating at  $500^{\circ}$

$V_f$  = volume of water filtered (l)

FOOTNOTES:

1. The aluminum foil squares can be best numbered with a ball-point pen. Although the ink will be burned off, the impression of the number will remain readable. It is a good idea to number the pieces of foil in several places.
2. When placing the wet filter back into the foil packet, place a small slip of paper under the filter to prevent it from sticking to the foil when drying.
3. Samples may be left at 101° C. for an indeterminate length of time. It may take more than one hour for the filters to dry to a constant weight.
4. All dried filters are very hygroscopic and will absorb moisture rapidly. To assure the best results, therefore, filters should be kept in the dessicator at all times except when weighing.

## Procedure 10

## TOTAL ALKALINITY

REAGENTS:

1. Sulfuric Acid 1.0 N  
dissolve 28.0 ml of concentrated  $\text{H}_2\text{SO}_4$  (36 N) in about 900 ml of deionized water. Bring to exactly 1 liter volume.
2. Sulfuric Acid 0.02 N  
dissolve 20.0 ml of 1.0 N  $\text{H}_2\text{SO}_4$  in about 900 ml of deionized water. Bring to exactly 1 liter volume.

PROCEDURE:

1. add 50.0 ml of water sample to small beaker with magnetic stir bar in place
2. insert calibrated pH probe and stir on magnetic stir plate
3. titrate with 0.02 N  $\text{H}_2\text{SO}_4$  until pH reaches 4.8
4. record ml of titrant used

CALCULATIONS:

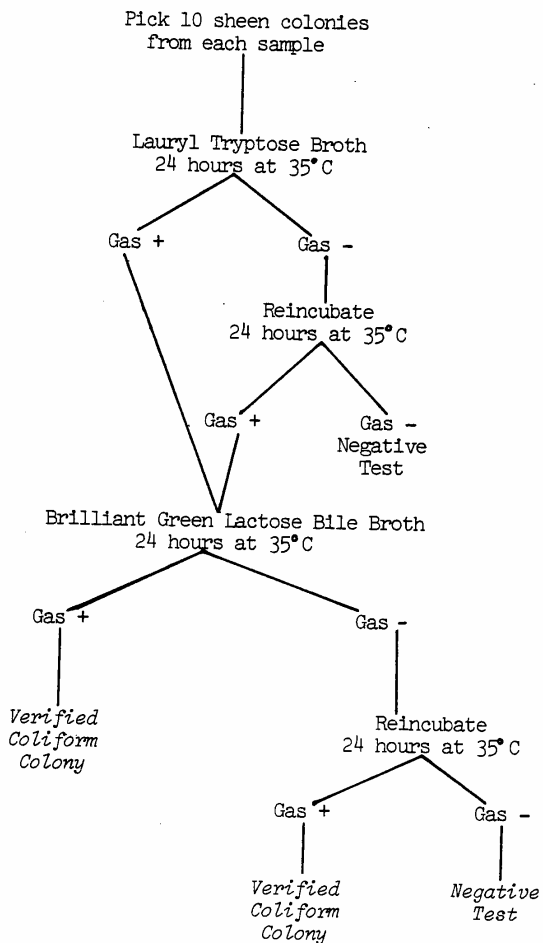
$$\text{Total Alkalinity (mg/l as CaCO}_3\text{)} = \frac{(A) \times (B \times 50,000)}{C}$$

where: A = total volume of acid titrant used  
B = normality of acid  
C = volume of sample (ml)

REFERENCES:

Wetzel, R.G. and G.E. Likens. 1979. Limnological analyses. W.B. Saunders Co. 357 pp.

TOTAL COLIFORMS



## FECAL COLIFORMS

Pick 10 blue colonies

Lauryl Tryptose Broth  
24 hours at 35°C

Gas +

Gas -

Reincubate  
24 hours at 35°C

Gas +

Gas -

EC Broth  
24 hours at 44.5°C

*Negative  
Test*

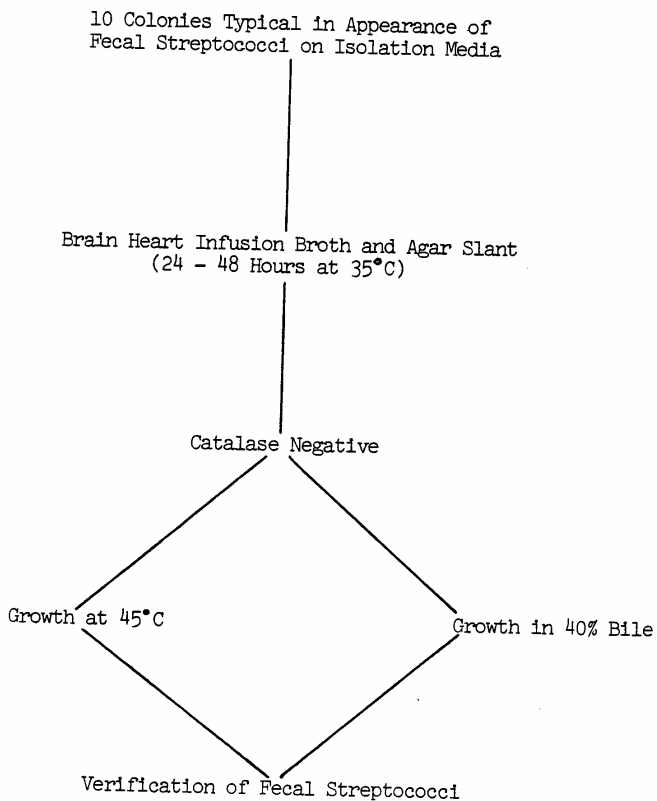
Gas +

Gas -

*Verified  
Fecal Coliform*

*Negative  
Test*

FECAL STREPTOCOCCI



## Procedure 12

CHLOROPHYLL a ANALYSIS

## PROCEDURE:

1. Filter algae onto 0.45 filters (either glass fiber or millipore filters). Record the volume filtered (ml).\*
2. Grind filter in 90% acetone. Transfer to a numbered centrifuge tube and add acetone to bring to volume. Record the volume of acetone (ml).
3. Place in the dark for 10 minutes to extract.
4. Shake the tube and place in the centrifuge. Centrifuge for 10 minutes.
5. Transfer the supernatant to a 1.0 cm spectrophotometer cell.
6. Read and record the optical density at 750, 665, 663, 645, and 630 mμ. At each wavelength the spectrophotometer must be zeroed using an acetone blank.
7. Add 2 drops of 1N HCl to sample and blank. Read and record the optical density at 750 and 665 mμ.
8. Calculate the amount of chlorophyll a and phaeo-pigments as follows:

## a) Scor-Unesco Trichromatic Equations

$$\text{chl } \underline{a} \text{ (ug/l)} = \frac{(11.64 \cdot E_{663}^b - 2.16 \cdot E_{645}^b + 0.10 \cdot E_{630}^b) \cdot v}{(V/1000 \cdot l)}$$

## b) Lorenzen's Equations

$$\text{chl } \underline{a} \text{ (ug/l)} = \frac{26.7 \cdot (E_{665}^b - E_{665}^a) \cdot v}{(V/1000) \times l}$$

$$\text{Phaeo-pigments (ug/l)} = \frac{26.7 \times (1.7 (E_{665}^a) - E_{665}^b) \times v}{(V/1000) \times l}$$

Where:  $E_{663}$ ,  $E_{665}$ ,  $E_{645}$ , and  $E_{630}$  are corrected absorbances  
(i.e. - after subtracting 750 mμ reading.

$v$  = volume of acetone (ml)

$V$  = volume of water filtered (ml)

$l$  = path length of cuvette (cm)

$a$  = after acid

$b$  = before acid

REFERENCES:

Lorenzen, C. J. 1967. Determination of chlorophyll and phaeopigments: Spectrophotometric Equations. *Limnol. and Oceanogr.* 12:335-338.

Strickland, J. D. H. and T. R. Parsons. 1972. A practical handbook of sea water and analysis. *Bull. of Fisheries Res. Board Can.* 167. 2nd ed. 310 pp.

- \* If you are unable to analyze the samples immediately, they may be stored at 4° C in the dark for up to a week without significant change in chlorophyll a. This should be checked out, however, for your own satisfaction. Freezing is not a good method of storage because it appears to destroy the porphyrin ring which causes significant changes in chlorophyll a concentrations.

## ALGAL ASSAY PROCEDURE

NOTE: Maintain aseptic conditions as much as possible throughout experiment. Autoclave equipment and make transfers and inoculations in sterile transfer room.

## EQUIPMENT AND GLASSWARE PREPARATION - PREPARE, AUTOCLAVE, AND LABEL

1. Appropriate number of 250 ml Erlenmeyer flasks.
2. Sample jugs.
3. E.P.A. Artificial Nutrient Media. (plus 4-6 250 ml flasks with 150 ml.)
4. E.P.A.  $K_2HPO_4$ -Free Nutrient Media.
5. (3) 250 ml Erlenmeyer flasks with 50 ml of  $K_2HPO_4$ -free media.
6. (3) 900 ml beakers.
7. (15-20) large (50 ml) centrifugation tubes.
8. 10 ml and 1 ml pipette tips.
9. Nutrient spikes A, B, & C.
10. (4-6) 50-100 ml beakers.
11. (2) 4l filter flasks.
12. (2) Gelman Filter Tops (with sterile filters).
13. (4-6) 1l flasks.
14. (4-5) 250 ml graduate cylinders.

MOTHER CULTURE PREPARATION

1. Transfer 7-10 day old cultures (4-6 150 ml E.P.A. culture flasks) into sterile 50 ml centrifuge tubes.
2. Centrifuge @ 2000 RPM for 1-2 minutes.
3. Decant down to 10 ml, resuspend pellet with vortex mixer in P-free media, and recentrifuge.
4. Decant, resuspend pellet, and pour into P-free media flask.
5. Rinse centrifuge tube with P-free media.
6. Starve for 24 hrs.

TEST WATER COLLECTION

1. Fill jugs with epilimnetic water, exclude air, cap, and chill.

TEST WATER PREPARATION

1. Pour in 1l flasks.
2. Autoclave at  $121^{\circ}C$  and 15 p.s.i. for 30 minutes. Remove and cool.
3. Filter.
4. Pour into beakers.

5. Pour into 250 ml graduate cylinder.
6. Pour into flasks (150 ml).
7. Allow to CO<sub>2</sub> equilibration for 24 hours. Shake.

TEST WATER NUTRIENT SPIKES

1. Autoclave nutrient spikes A, B & C.
2. Add appropriate spike volume to respective flasks.

INOCULUM PREPARATION

1. Count P-starved mother culture (triplicates of different volumes).
2. Adjust mother culture to correct density so that 1-2 ml of mother culture diluted to 150 ml yields ca 1000 cells ml<sup>-1</sup>.
3. Add appropriate volume to each flask (Test & Controls).
4. Incubate for 14 days, shaking once per day.

BIOMASS MEASUREMENT

1. Gravimetric (Method II)
  - A. Dry appropriate number of filters (0.45  $\mu$ m Millipore-Type BD) for 2 hours @ 70° C (not more than 75° C).
  - B. Cool filters in dessicator for 1 hour before weighing.
  - C. Filter a suitable measured aliquot of culture.
  - D. Rinse the filter funnel with distilled H<sub>2</sub>O.
  - E. Dry the filter to constant weight @ 70° C, cool for 1 hour in a desiccator, obtain weight.
2. Cell Counts
  - A. Count a measured aliquot of culture (triplicates of different volumes).

## Procedure 13 (con't.)

NUTRIENT ADDITION  
EXPERIMENTAL DESIGN

<u>Treatment</u>	<u>Conc.</u>	<u>Flask</u>	<u>Conc.</u>	<u>Additions</u>
Lake H <sub>2</sub> O + 1	0.05 mg P/l	0.0075 mg/150 ml		Add 1 ml of Sol. A.
Lake H <sub>2</sub> O + 2	1.00 mg N/l	0.15 mg/150 ml		Add 1 ml of Sol. B.
Lake H <sub>2</sub> O + 3	0.05 mg P/l + 1.00 mg N/l	.0075 mg/150 ml + 0.15 mg/150 ml		Add 1 ml of Sol. A. + Add 1 ml of Sol. B.
Lake H <sub>2</sub> O + 4	1.00 mg Na <sub>2</sub> EDTA/l	0.15 mg/150 ml		Add 1 ml of Sol. C.
Lake H <sub>2</sub> O + 5	0.05 mg P/l + 1.00 mg Na <sub>2</sub> EDTA/l	0.0075 mg/150 ml + 0.15 mg/150 ml		Add 1 ml of Sol. A. + Add 1 ml of Sol. C.
Lake H <sub>2</sub> O + 6	1.00 mg N/l + 1.00 mg Na <sub>2</sub> EDTA/l	0.15 mg/150 ml + 0.15 mg/150 ml		Add 1 ml of Sol. B. + Add 1 ml of Sol. C.
Lake H <sub>2</sub> O + 7	0.05 mg P/l + 1.00 mg N/l + 1.00 mg Na <sub>2</sub> EDTA/l	0.0075 mg/150 ml + 0.15 mg/150 ml + 0.15 mg/150 ml		Add 1 ml of Sol. A. + Add 1 ml of Sol. B. + Add 1 ml of Sol. C.
8	Synthetic Algal Nutrient Medium Control			
9	Test Lake Water Control			

Sol. A. is 0.0075 mg P/ml K<sub>2</sub>HPO<sub>4</sub>Sol. B. is 0.15 mg N/ml NaNO<sub>3</sub>Sol. C. is 0.15 mg Na<sub>2</sub> EDTA/ml Na<sub>2</sub> EDTA

## TOTAL PHOSPHORUS (MACROPHYTE)

REAGENTS

1. See procedure 2a.

METHOD

1. Grind the dried plant tissue with a mortar and pestle.
2. Place it into a marked vial and return to drying oven for 24 hours.
3. Place 100-150 mg of sample into 125 ml flask containing 50 ml deionized water. Record exact weight used on label.
4. Add 7.5 ml of potassium persulfate digest solutions (wash down the particles from the side of the flask with the addition).
5. Autoclave for 1 hour at 15 p.s.i. and 121° C.
6. Remove and cool to room temperature.
7. Carefully, avoiding particulate material, remove 20 ml of supernatant with an autopipet and place into a 50 ml beaker.
8. Carefully remove small amount of clear liquid and read on a spectrophotometer at 880 nm in a 1 cm cell for background turbidity level (absorbance) and record.
9. Adjust pH of a 20 ml sample removed from the original flask by first adding 2.0 drops of phenolphthalein indicator.
10. Add 10 N NaOH Dropwise until indicator turns pink.
11. Add 1 N HCl Dropwise until pink color just disappears.
12. Add 5.0 ml of phosphate color reagent and allow at least 10 minutes for color development (color is stable for 1 hour).
13. Read at 880 nm against a deionized water blank and record (a blank is prepared exactly as a sample except that no phosphate reagent is added).

STANDARD PREPARATION

1. See procedure 2b.

# TOTAL NITROGEN (MACROPHYTE)

## REAGENTS

1. See procedure 6.

## METHOD

1. Grind dried plant tissue with mortar and pestle.
2. Place into marked vial and return to drying oven for 24 hours.
3. Place 100-150 mg into 125 ml flask containing 50 ml deionized water (Record exact weight used on label).
4. Carefully add exactly 25.0 ml deionized water. Avoid getting plant tissue on the side of the flask.
5. Add 1.0 g of potassium persulfate.
6. Autoclave for 1 hour at 15 p.s.i. and 121° C.
7. Remove and cool to room temperature and add 0.5 g of Devarda's alloy.
8. Allow to stand for 24 hours.
9. Place approximately 10.0 ml of sample liquid in a small centrifuge tube and centrifuge for 3 minutes at approximately 2000 r.p.m.
10. Carefully remove 1 ml of supernatant with an automatic pipette.
11. Place a 1 ml subsample into 24 ml deionized (1 in 25 dil.). Add 3 drops of 3N NaOH to each flask. Check pH. The pH should be approximately 5.0. If it isn't, adjust dropwise with 1N NaOH and swirl.
12. Add 1.0 ml of Solorzano phenol sol., 1.0 ml of sodium nitroprusside, and 2.5 ml of oxidizing reagent to each beaker.
13. Color development can be read spectrophotometrically after 1 hour at 640 nm.

## STANDARD PREPARATION

1. See procedure 6.

## SEDIMENT ANALYSIS

FIELD PROCEDURES

1. Collect sediment and interstitial water in acid-cleaned glass jars using scuba techniques.
2. Immediately add 1.5 ml 10.8 N  $\text{H}_2\text{SO}_4$  to samples collected for nutrient analysis.
  - A. Do not add acid to samples collected for gravimetric analysis of moisture, inorganic, and organic content.
3. Place samples on ice for transportation to laboratory.

Laboratory Procedures

## ANALYSIS OF TOTAL PHOSPHORUS IN INTERSTITIAL WATER

1. Allow acidified sediment samples to stand overnight under refrigeration to allow particulate material to settle out.
2. Carefully remove 50.0 ml of the overlying water with a 50 ml volumetric pipette - do not disturb sediments!
3. Carry out total P analysis using the persulfate digestion method with the following modifications:
  - A. After autoclaving the samples, allow them to stand unagitated during cooling and carefully remove 20.0 ml of supernatant with autopipette (avoid particulate matter accumulated on bottom of flask).
  - B. Transfer 20.0 ml subsample to 50 ml beaker, adjust pH, and add 2.0 ml phosphate color reagent - after 10 minutes, analyze spectrophotometrically at 880 nm.
  - C. Triplicate selected samples for statistical analysis.

## ANALYSIS OF ACID-NONLABILE SEDIMENT BOUND PHOSPHORUS

1. Decant overlying water and homogenize sediments with spatula.
2. Place portion in crucible and dry in drying oven at  $101^\circ\text{C}$  for 24 h.
3. Prior to removal, grind each dried sample with mortar and pestle to obtain a homogeneous powder.
4. Replace in drying oven for 2 h to ensure complete dehydration.
5. Remove subsample of 100 to 150 mg dry sediment (record exact weight!) and add to 125 ml flask containing 50.0 ml deionized water.
6. Carefully shake flask to wet sediments completely.
7. Slowly add 7.5 ml potassium persulfate solution, washing sediment particles from inside of flask into water.
8. Digest in autoclave for 1 h.

## Procedure 15 (con't)

9. Allow samples to cool without mixing, and carefully remove 20.0 ml of supernatant with a 10 ml autopipett (do not disturb sediments at bottom of flask!!).
10. Adjust pH, add 2.0 ml color reagent and proceed with spectrophotometric analysis.
11. Calculate  $\mu\text{g P (g dry weight)}^{-1}$  of sediment using this equation:

$$\mu\text{g P g}^{-1} = \frac{(A \times 20 \times 30.97)}{(B \times 20)}$$

where A is the value from standard curve for 50 ml sample (in  $\mu\text{Moles}$ ).

where B is the dry weight (g) of sediment digested.

12. Triplicate selected samples for statistical analysis.

ANALYSIS OF MOISTURE, INORGANIC, AND ORGANIC CONTENT OF SEDIMENT

1. Place 30 to 40 mls ( $=\text{cm}^3$ ) of unacidified wet sediment into tared crucible and obtain wet weight (expressed as (g wet sediment)  $\text{cm}^{-3}$ ).\*
2. Dry sample at  $101^\circ\text{C}$  for 24 h and obtain dry weight (expressed as (g dry sediment)  $\text{cm}^{-3}$ ).
3. Ash at  $550^\circ\text{C}$  for 2 h and obtain ash weight.
4. Calculate percent moisture, inorganic, and organic content per unit volume of sediment.

\*Note: Observe and record sediment texture and type at this point.

APPENDIX II  
TRIBUTARY MONITORING

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Fig. 1. Temporal variation of turbidity  
in the tributaries and Outlet of  
Lake of the Woods. Numbers denote  
sampling sites.

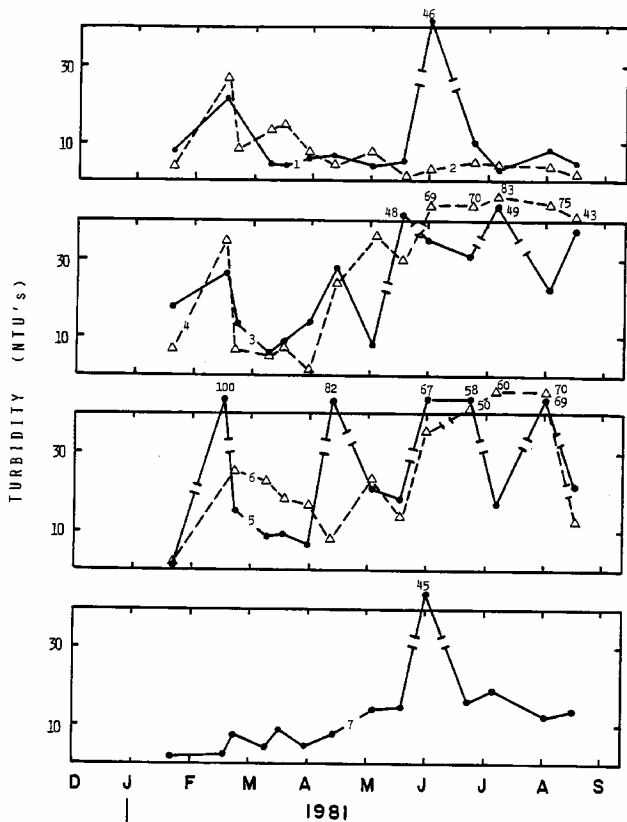


Fig. 2. Temporal variation of total phosphorus in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

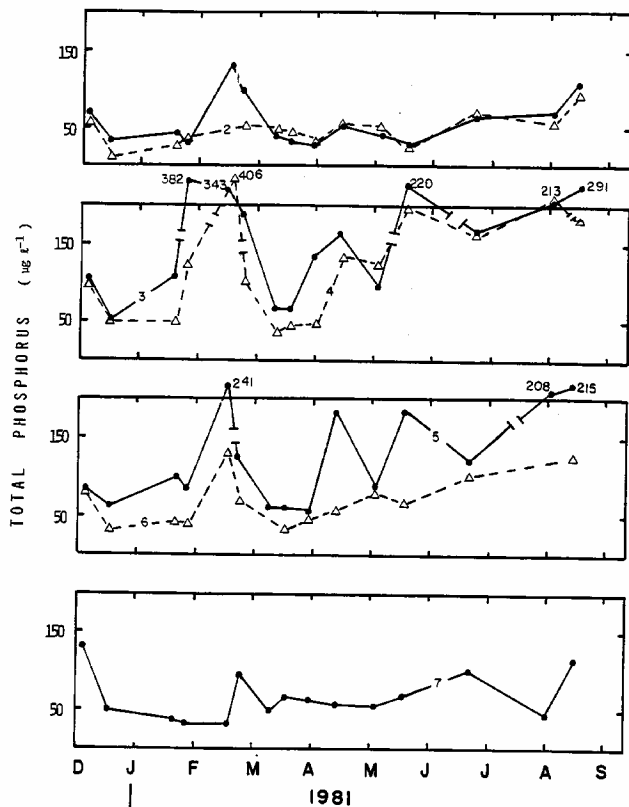


Fig. 3. Temporal variation of soluble reactive phosphorus in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

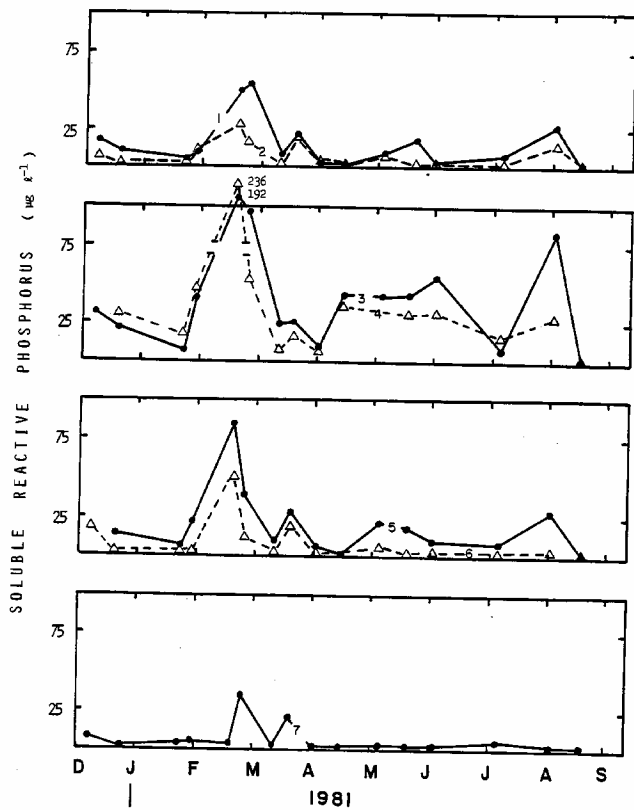


Fig. 4. Temporal variation of ammonia in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

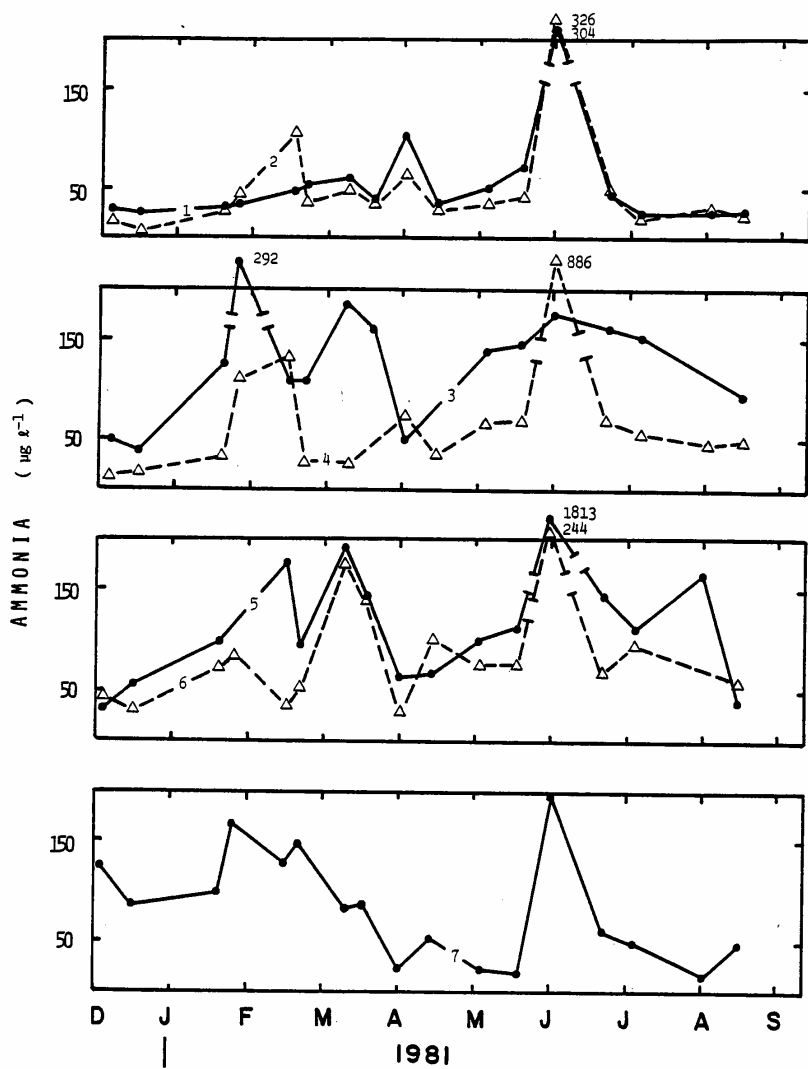


Fig. 5. Temporal variation of nitrite  
in the tributaries and Outlet  
of Lake of the Woods. Numbers  
denote sampling sites.

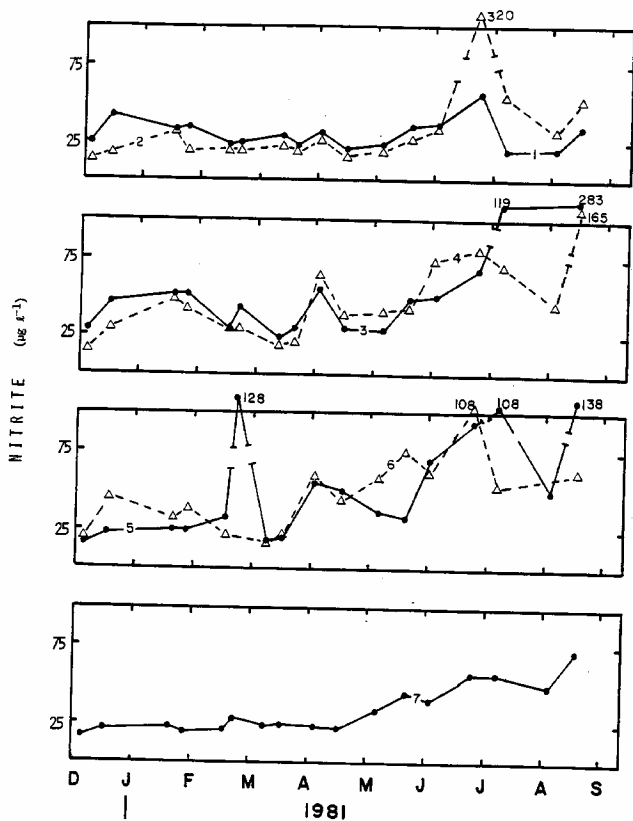


Fig. 6. Temporal variation of nitrate in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

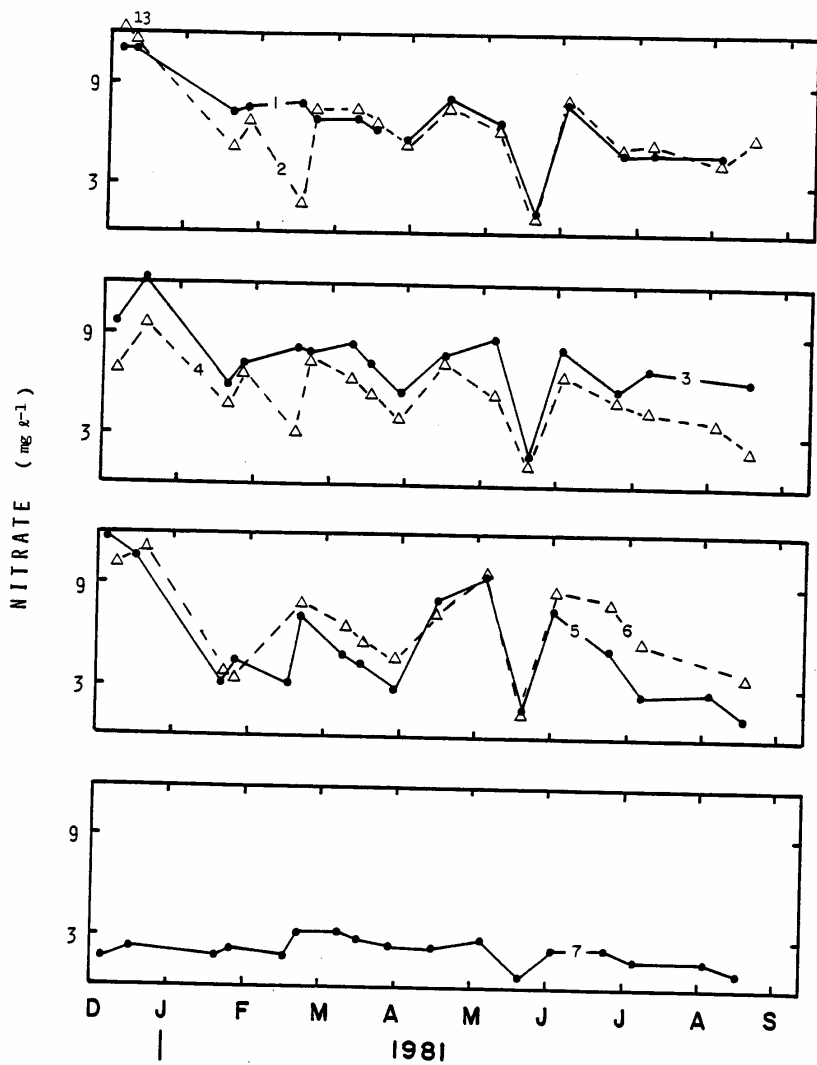


Fig. 7. Temporal variation of organic nitrogen in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

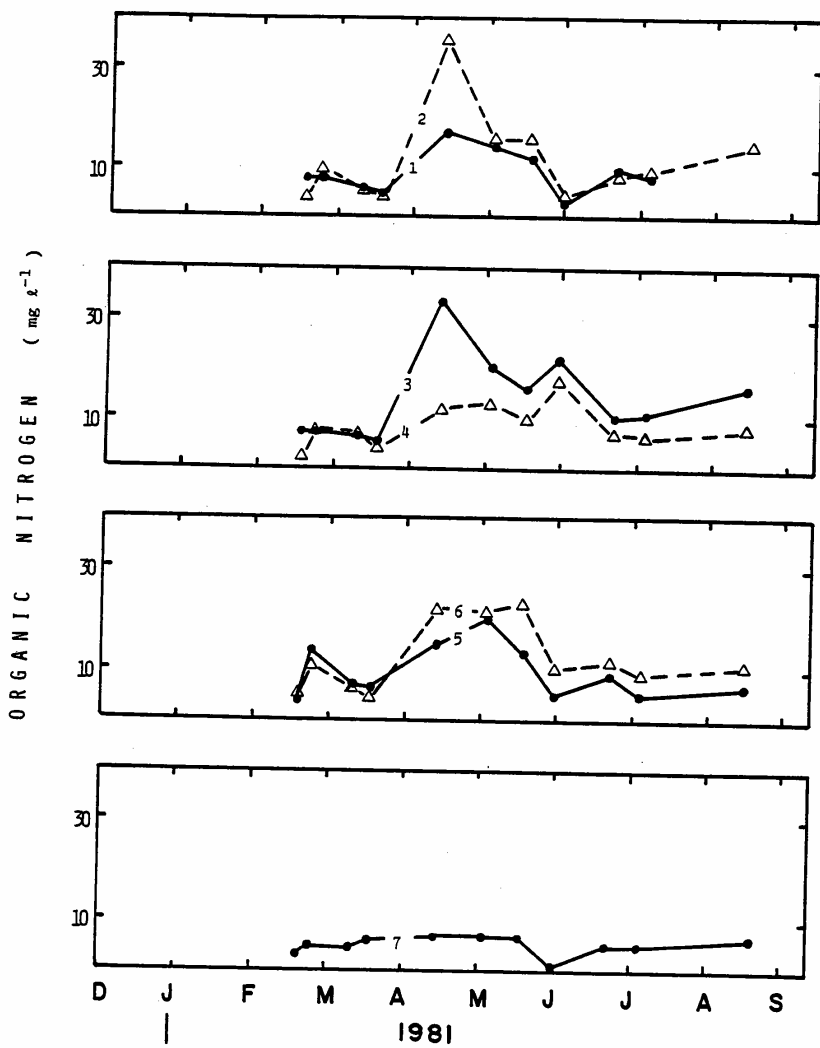


Fig. 8. Temporal variation of silica in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

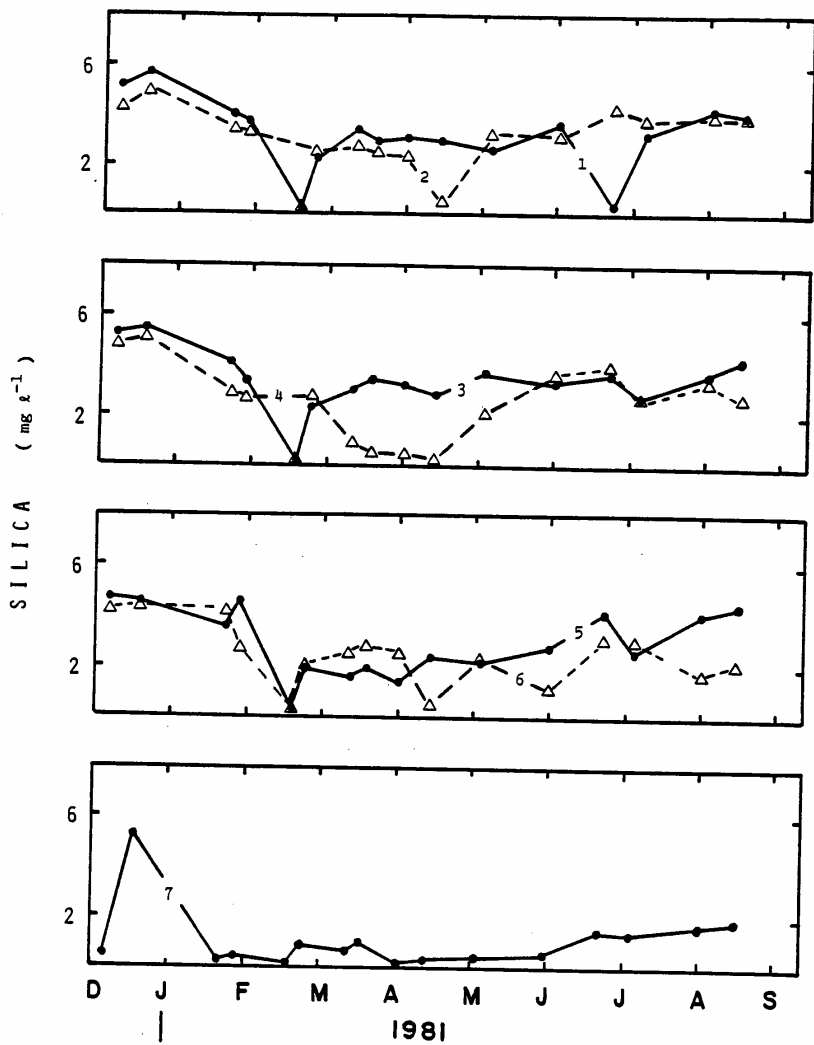


Fig. 9. Temporal variation of total  
residue in the tributaries and  
Outlet of Lake of the Woods.  
Numbers denote sampling sites.

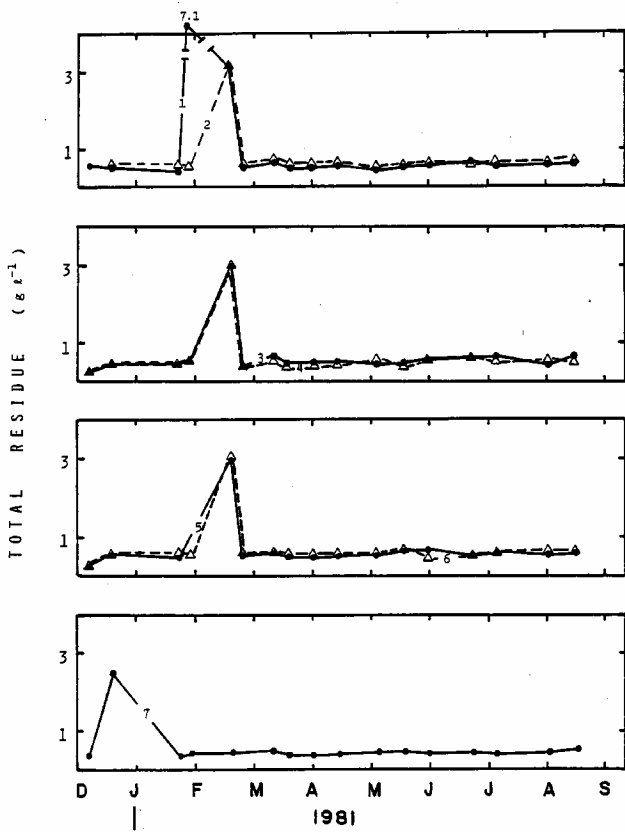


Fig. 10. Temporal variation of organic residue in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

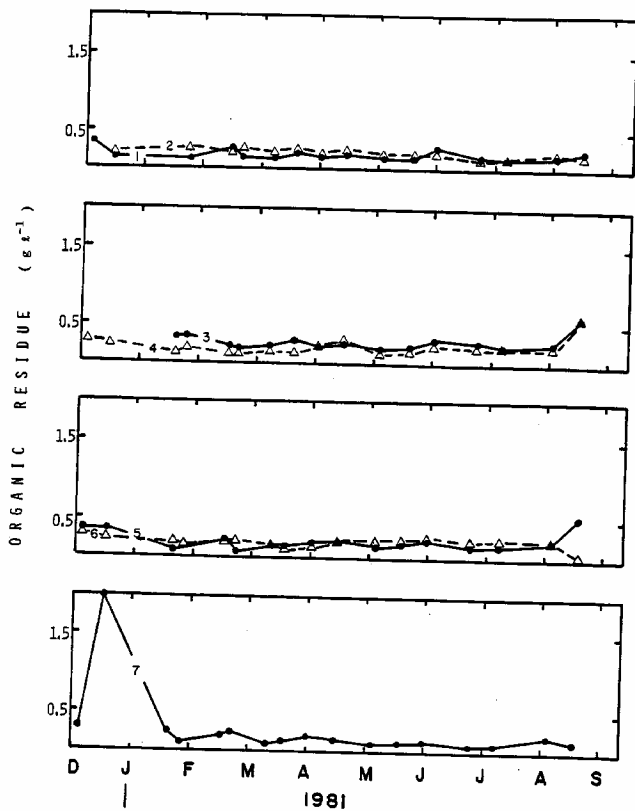


Fig. 11. Temporal variation of total particulate matter in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

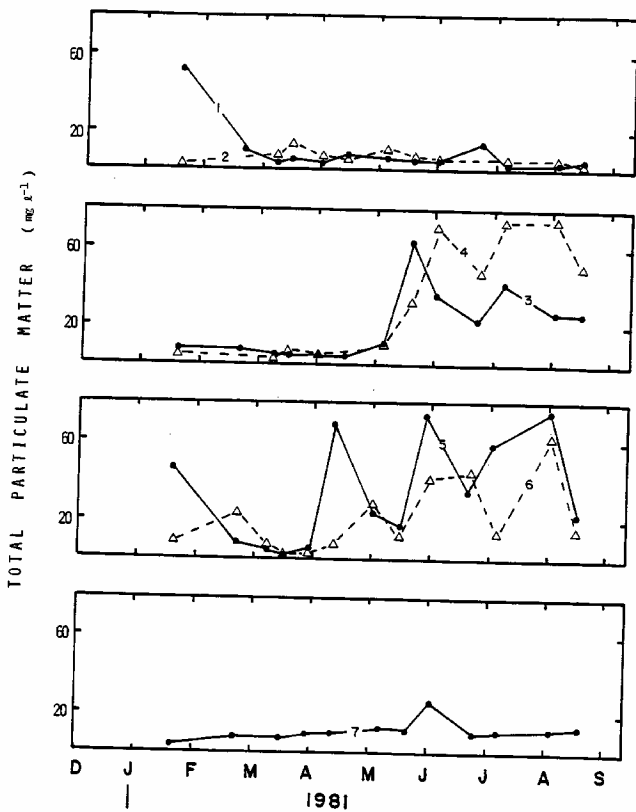


Fig. 12. Temporal variation of particulate organic matter in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.

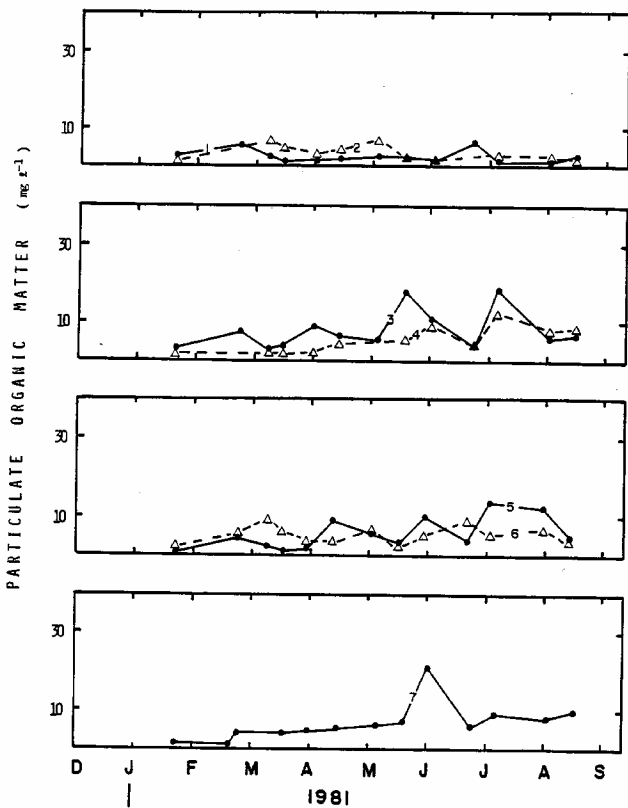
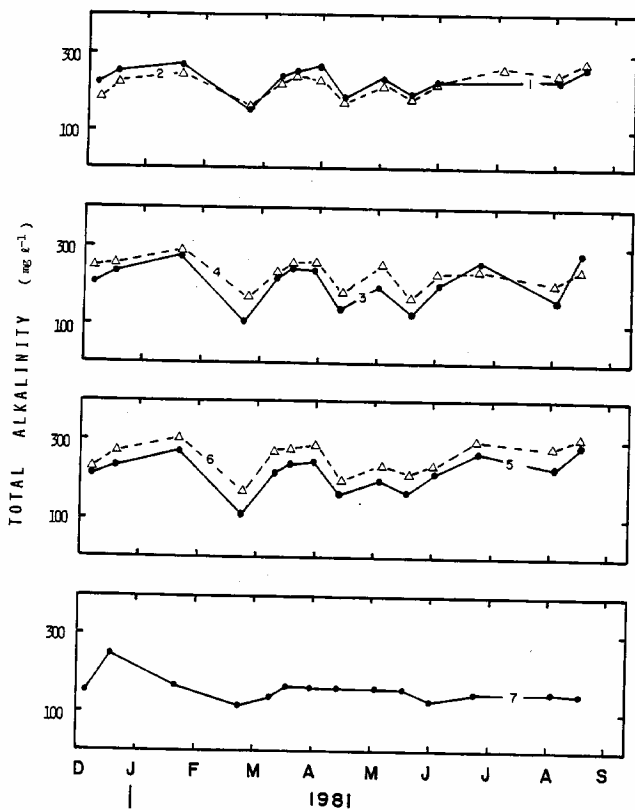


Fig. 13. Temporal variation of total alkalinity in the tributaries and Outlet of Lake of the Woods. Numbers denote sampling sites.





# APPENDIX III

## LAKE MONITORING

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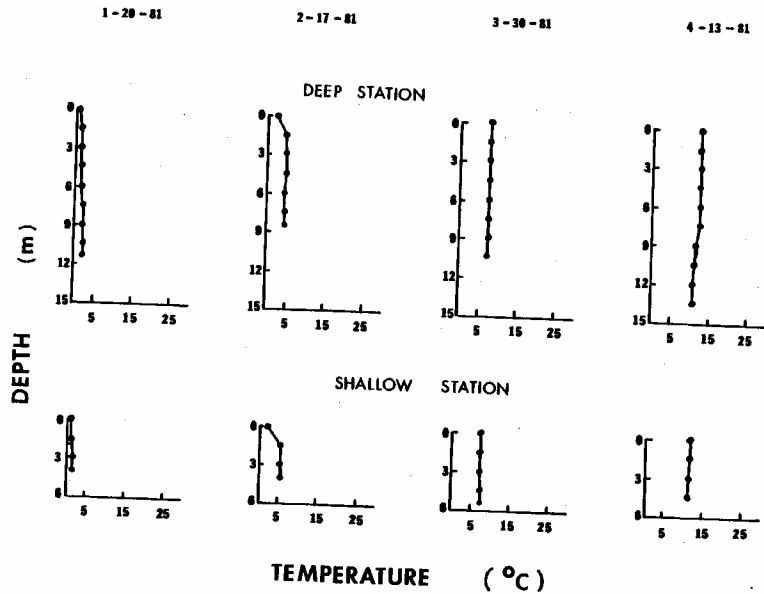
## APPENDIX III (con't.)

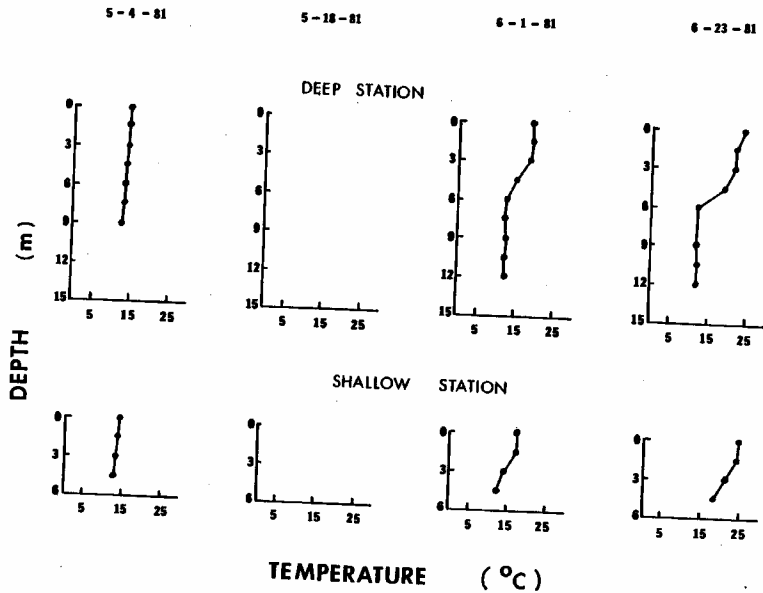
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Fig. 1a-c. Temporal variation in temperature profiles at the deep and shallow stations of Lake of the woods.





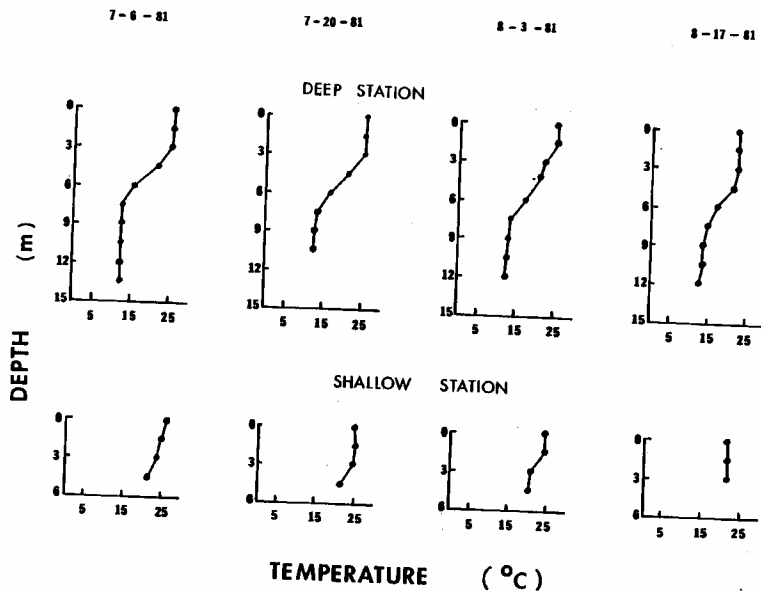
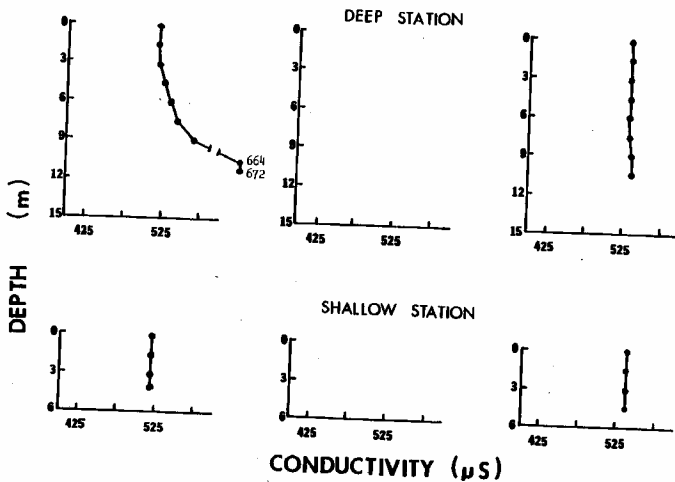


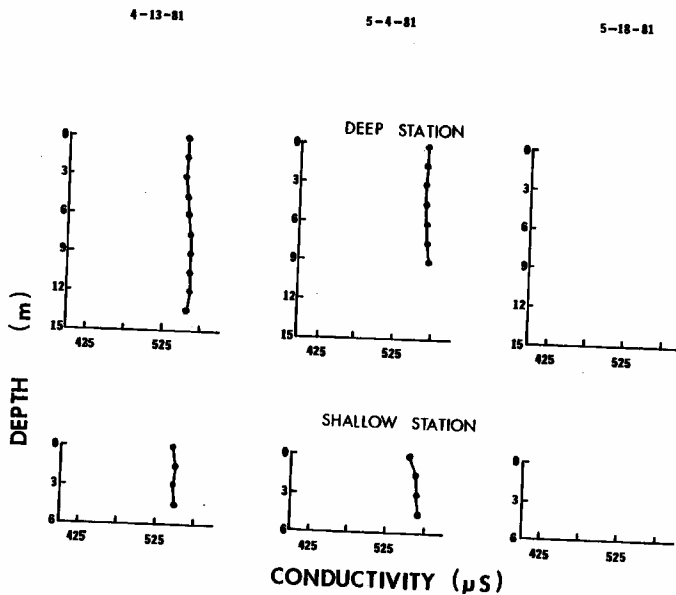
Fig. 2a-d. Temporal variation in conductivity profiles at the deep and shallow stations of Lake of the Woods.

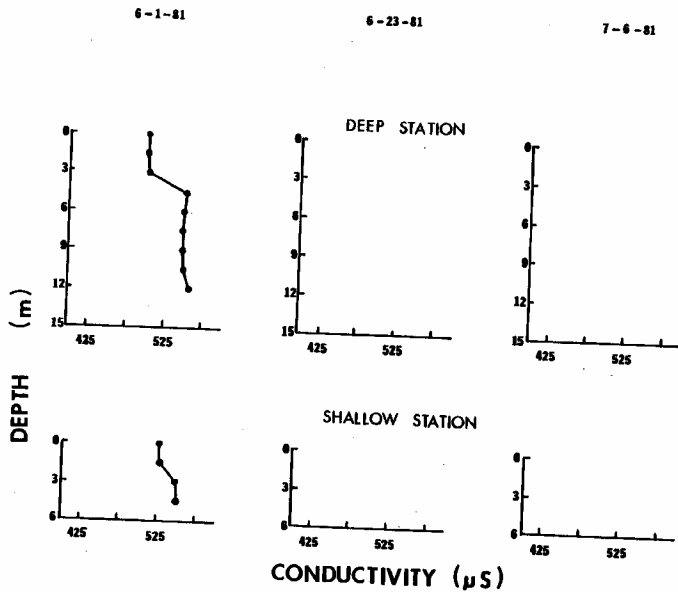
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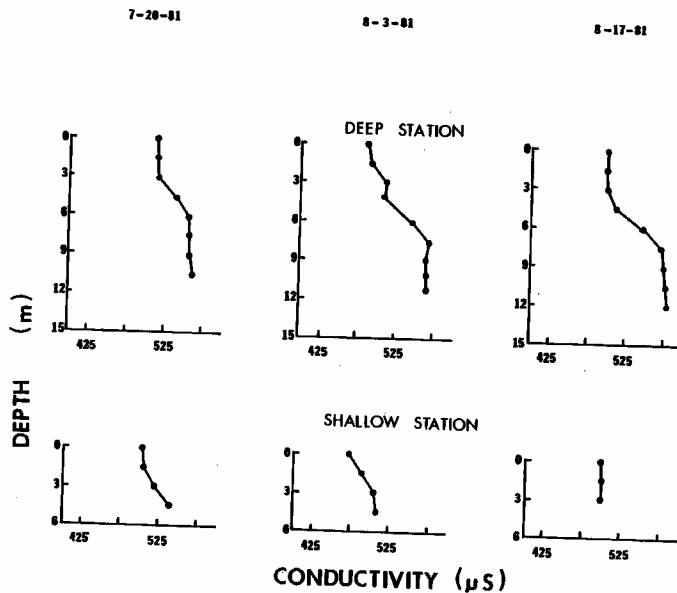
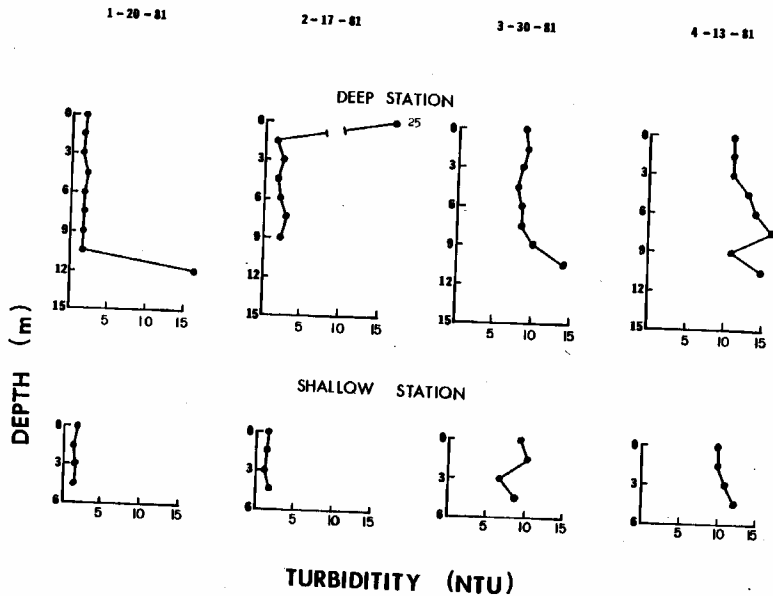
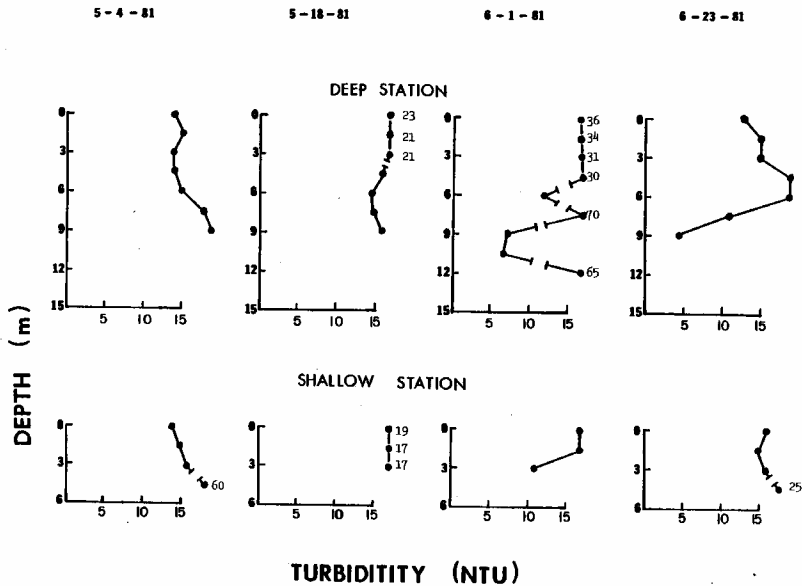


Fig. 2d



Fig. 3a-c. Temporal variation in turbidity profiles at the deep and shallow stations of Lake of the Woods.





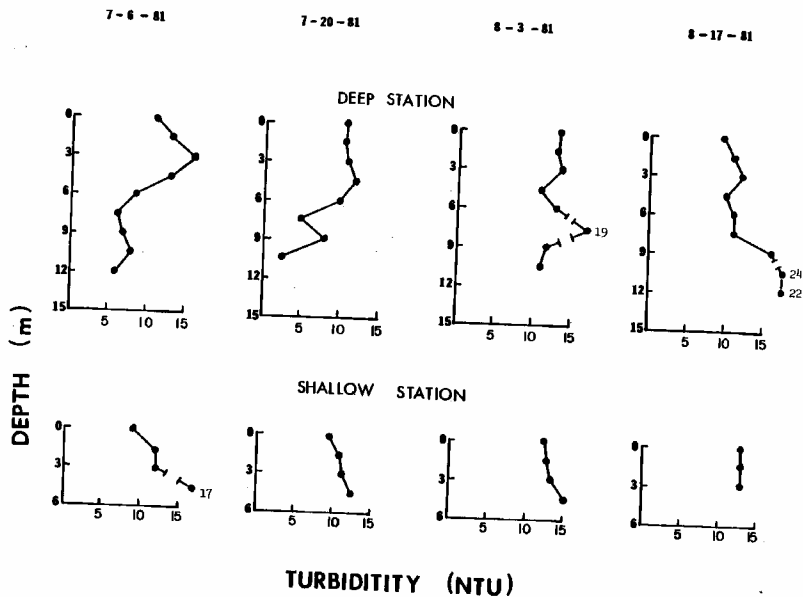


Fig. 4. Temporal variation in Secchi depth at the deep (●) and shallow (Δ) stations of Lake of the Woods.

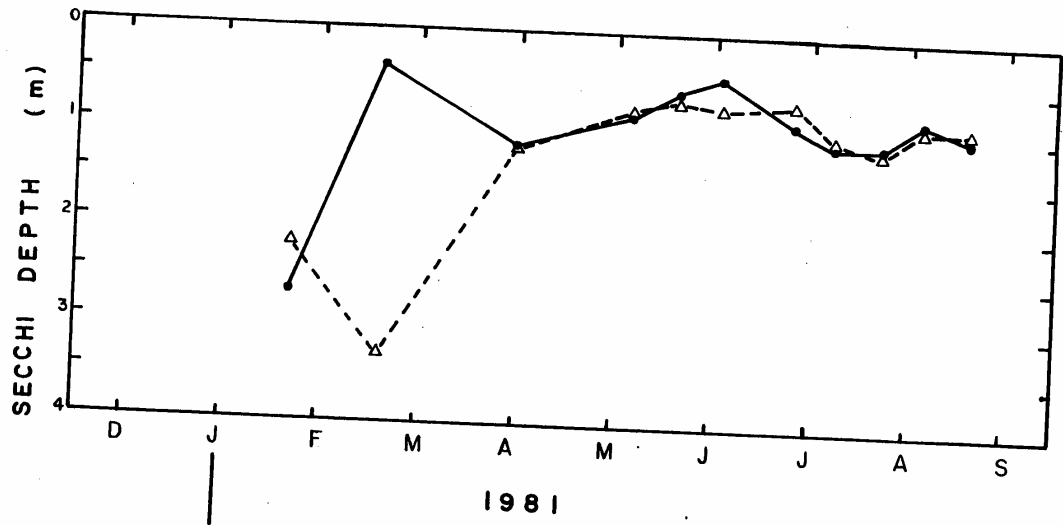
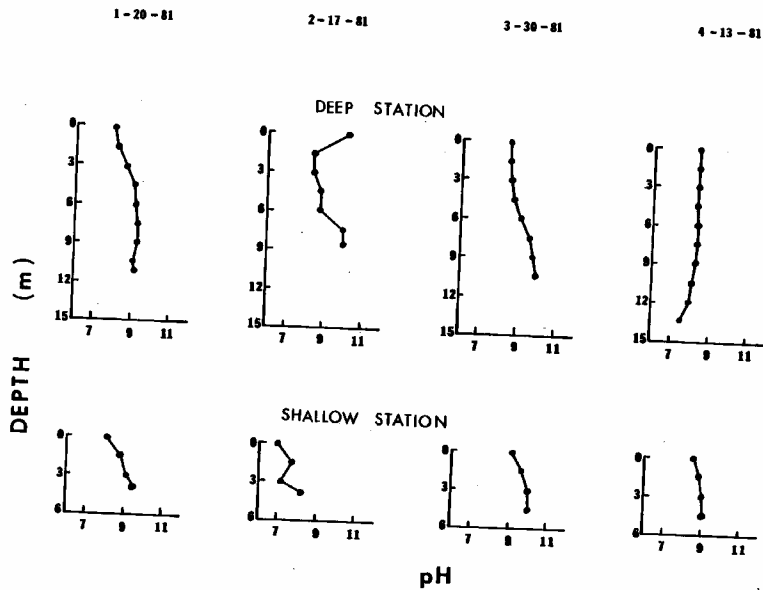
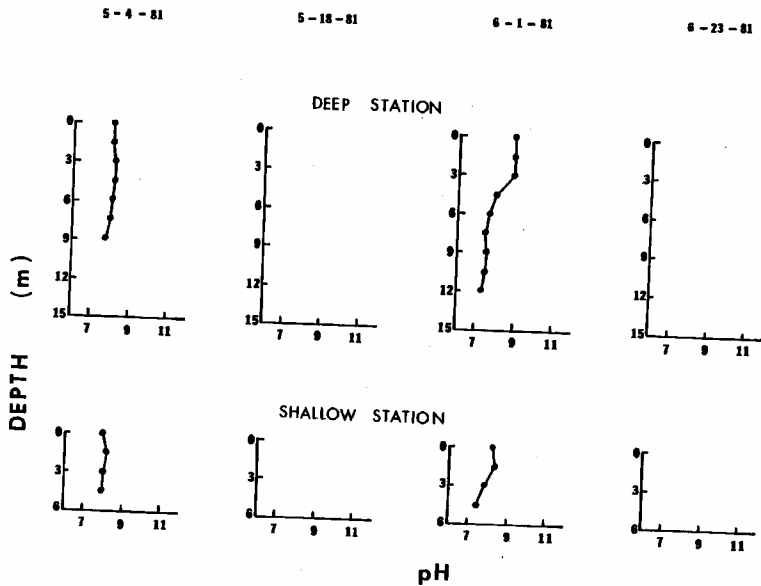


Fig. 4

Fig. 5a-c. Temporal variation in pH  
profiles at the deep and shallow  
stations of Lake of the Woods.





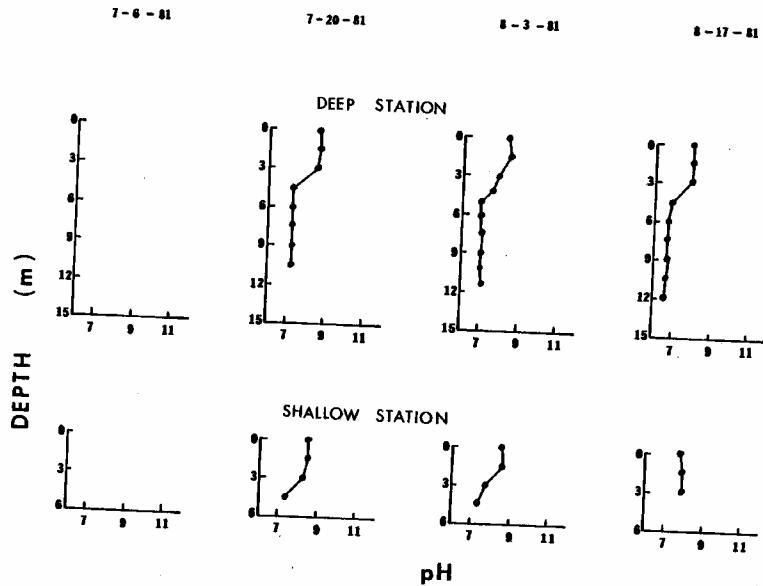
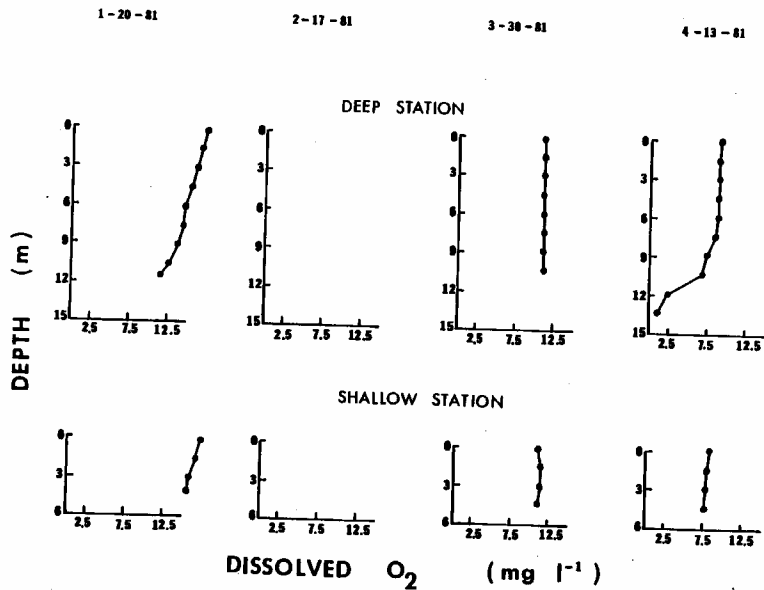
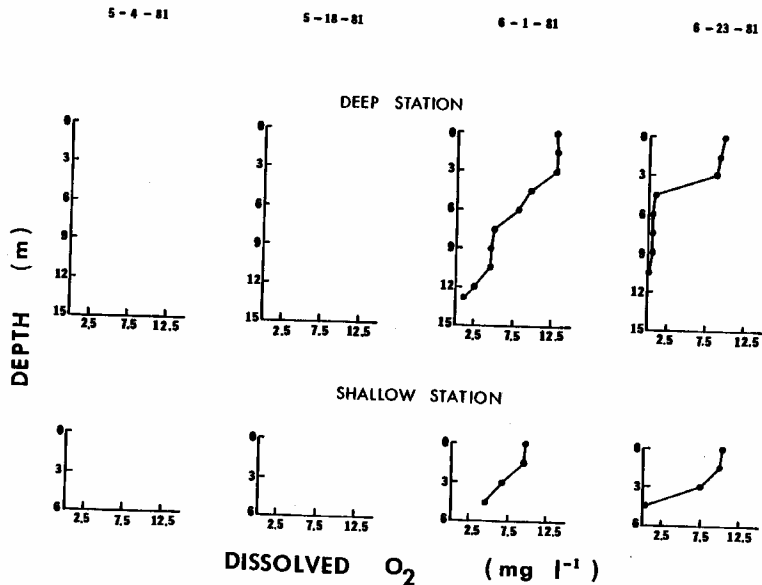


Fig. 6a-c. Temporal variation in dissolved oxygen profiles at the deep and shallow stations of Lake of the Woods.





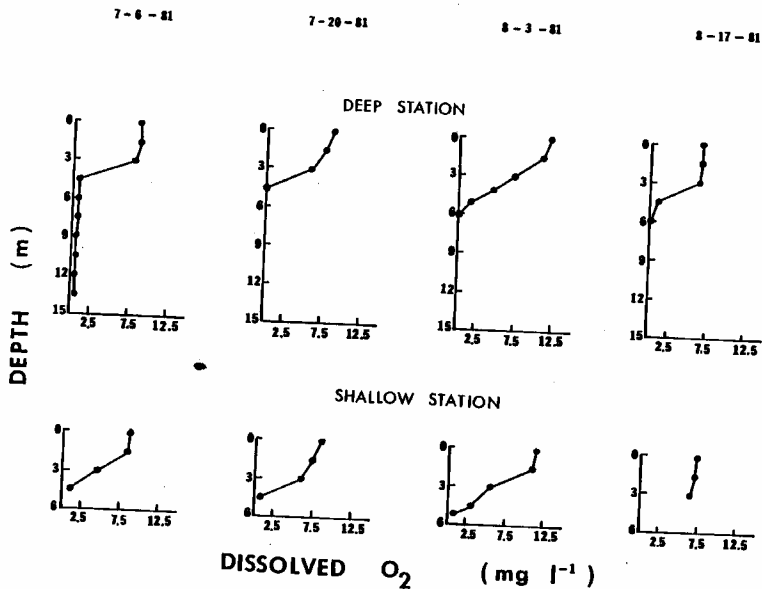
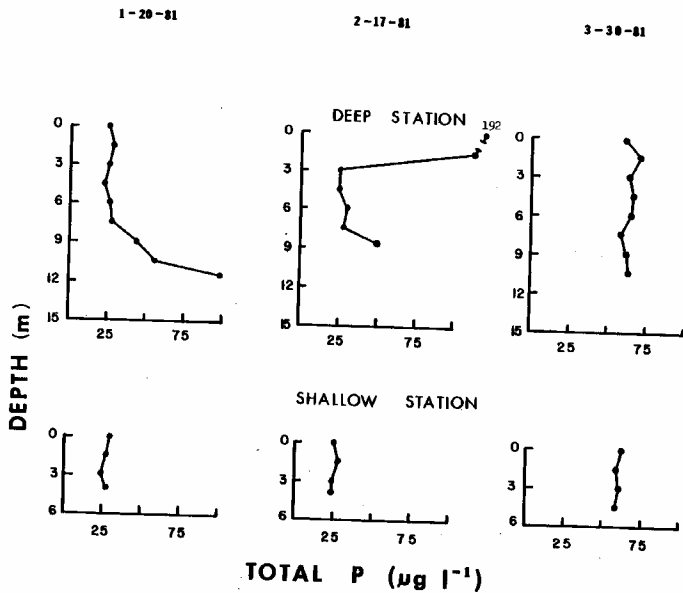
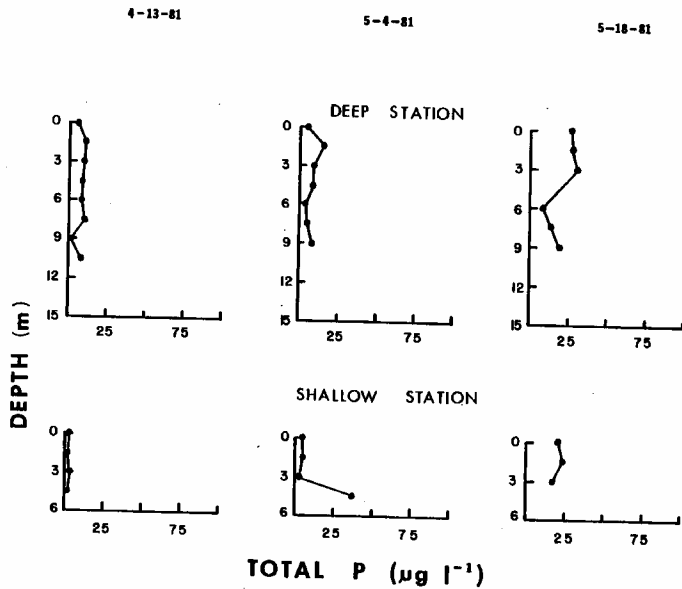
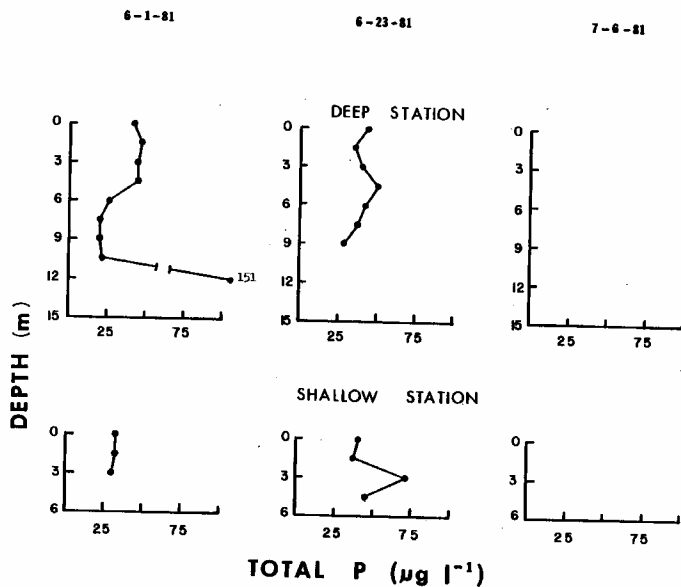


Fig. 7a-d. Temporal variation in total phosphorus profiles at the deep and shallow stations of Lake of the Woods.







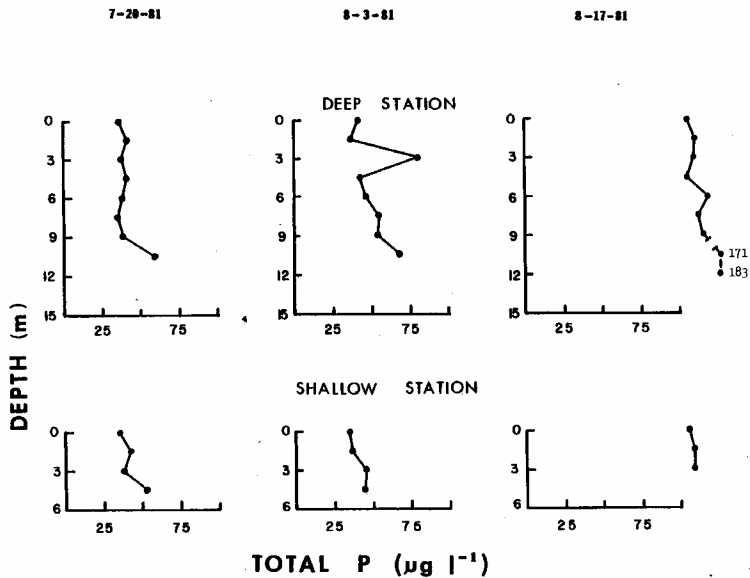
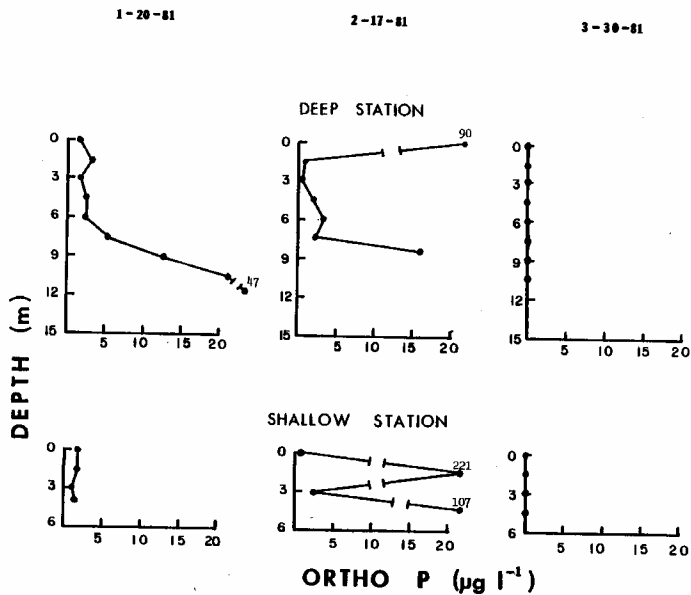
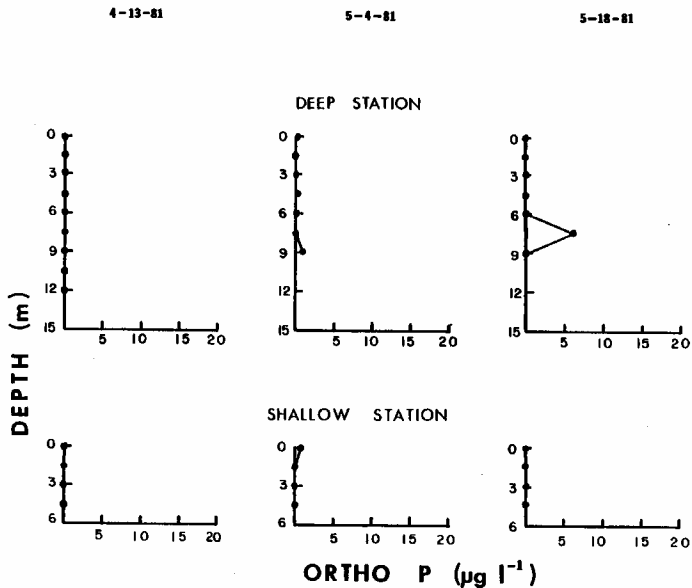




Fig. 8a-d. Temporal variation in soluble  
reactive phosphorus profiles  
at the deep and shallow stations  
of Lake of the Woods.

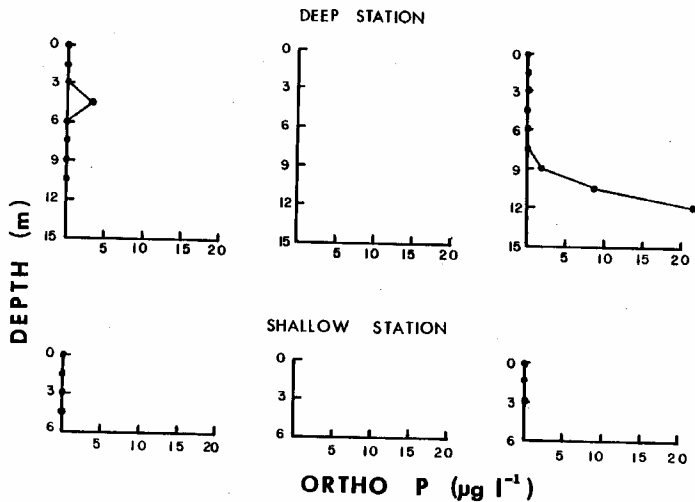




6-1-81

6-23-81

7-6-81



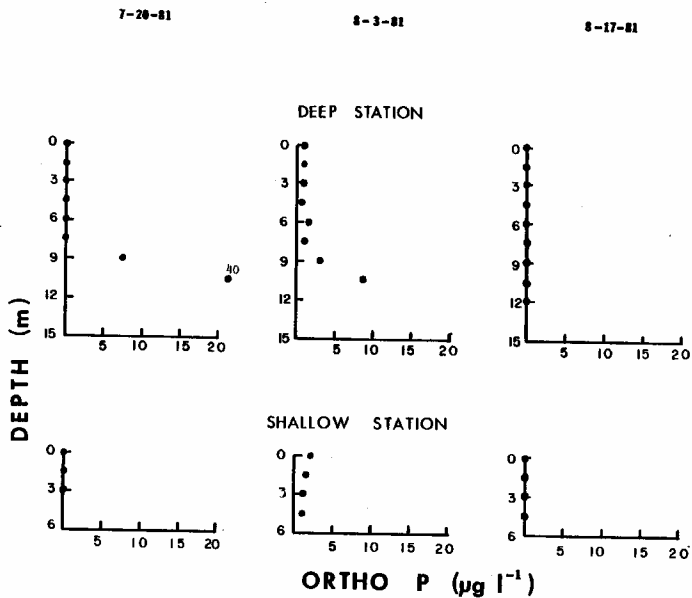
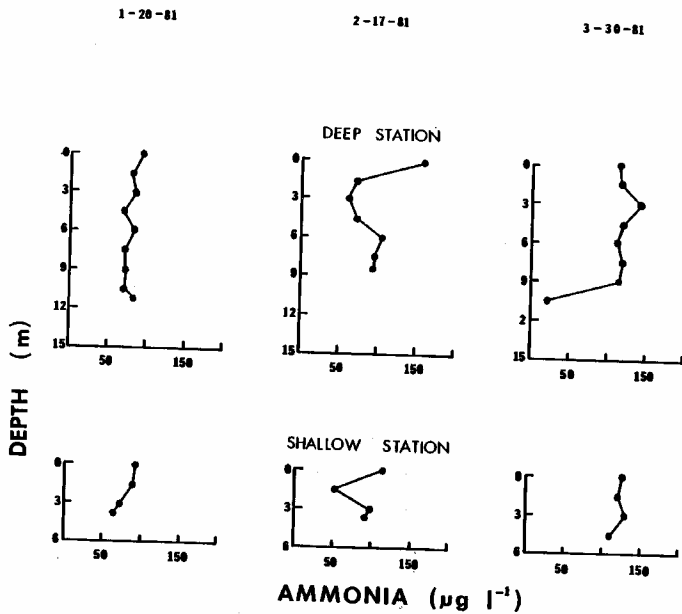
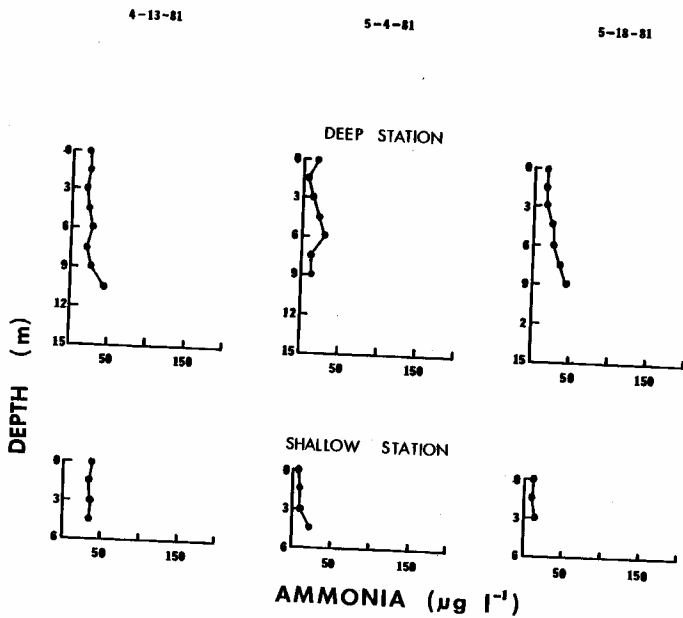
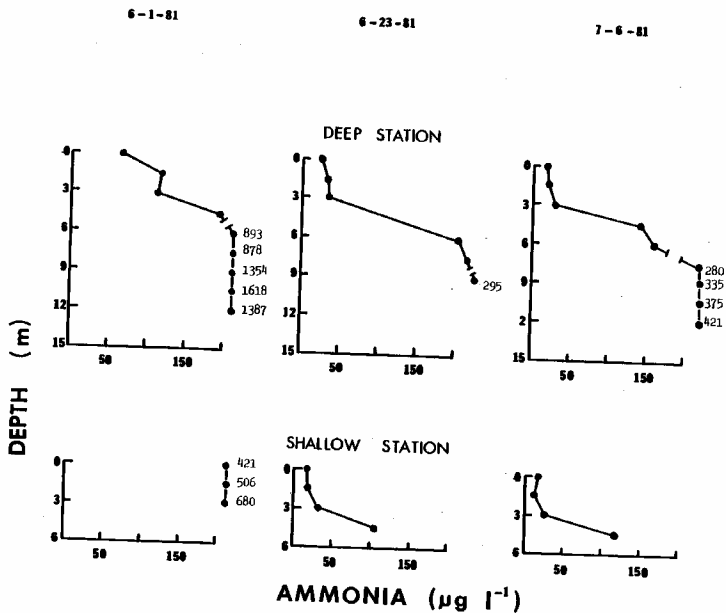




Fig. 9a-d. Temporal variation in ammonia profiles  
at the deep and shallow stations  
of Lake of the Woods.







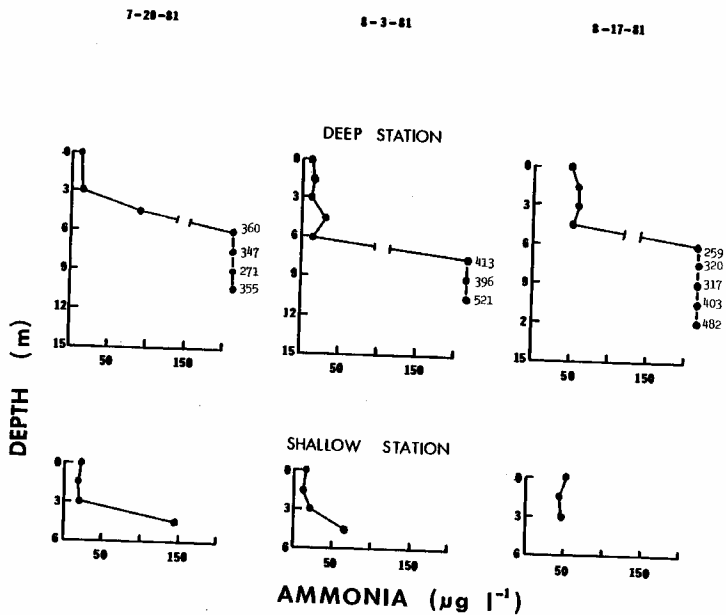


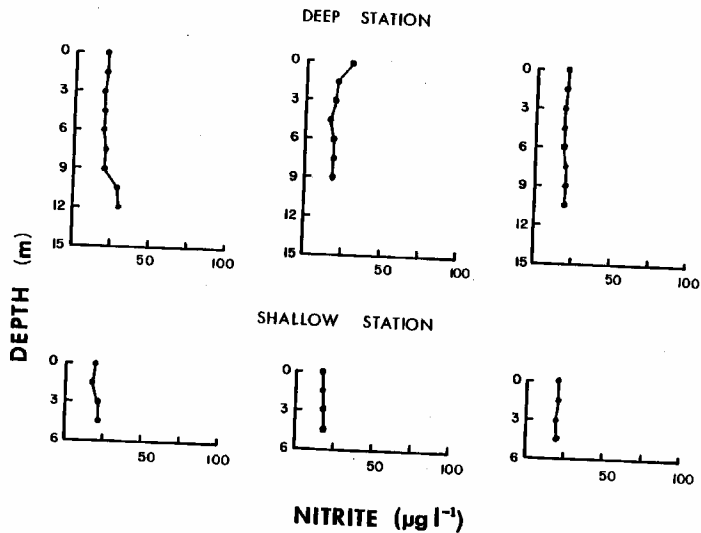


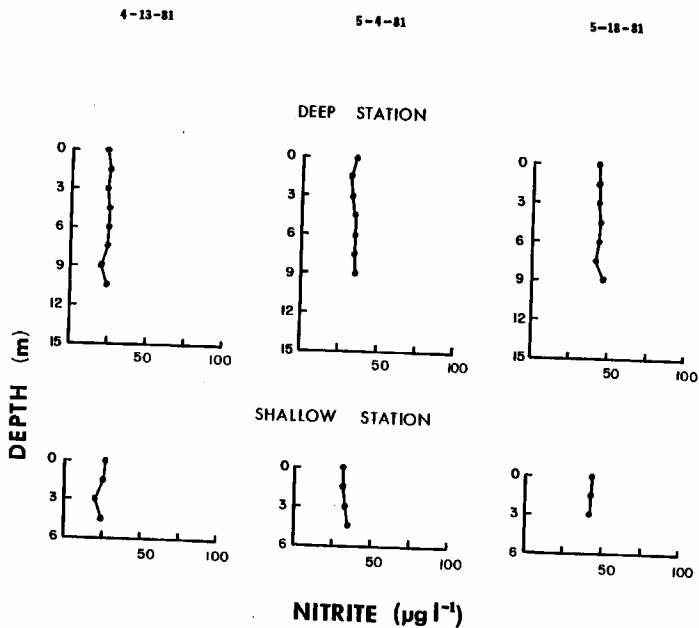
Fig. 10a-d. Temporal variation in nitrite profiles at the deep and shallow stations of Lake of the Woods.

1-20-81

2-17-81

3-30-81

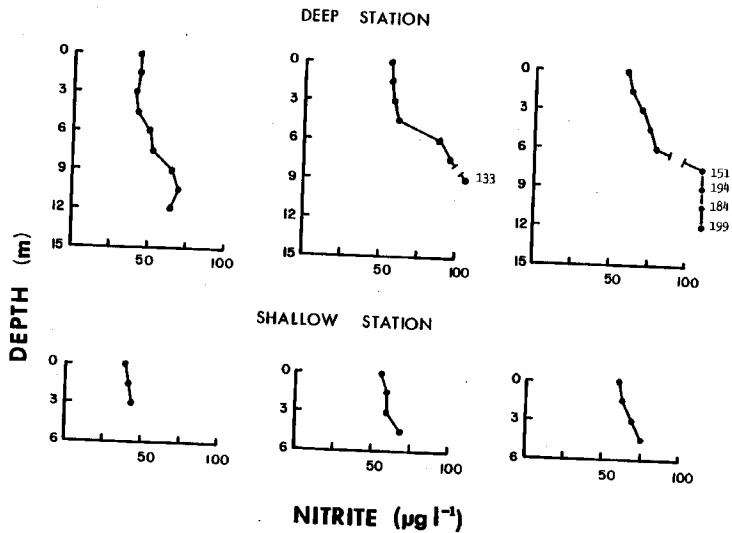




6-1-81

6-23-81

7-6-81



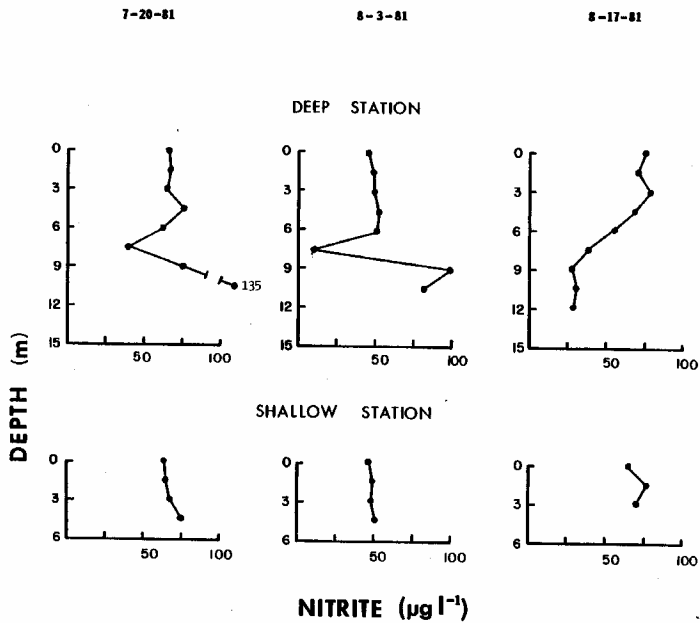
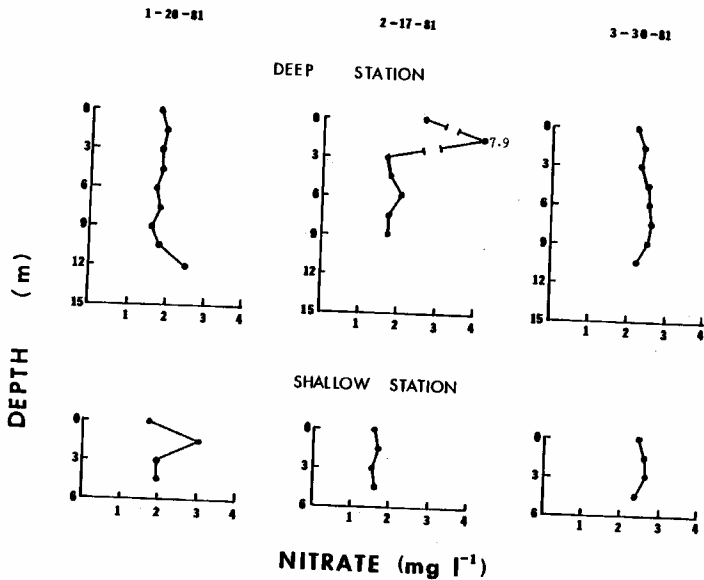
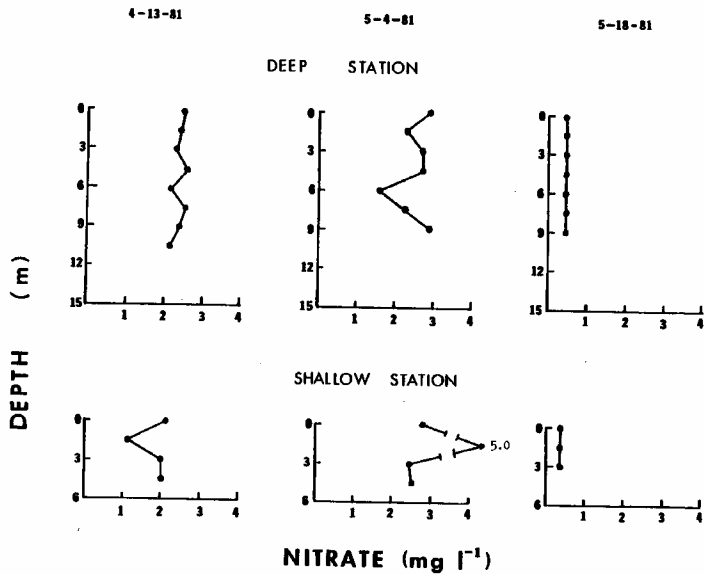
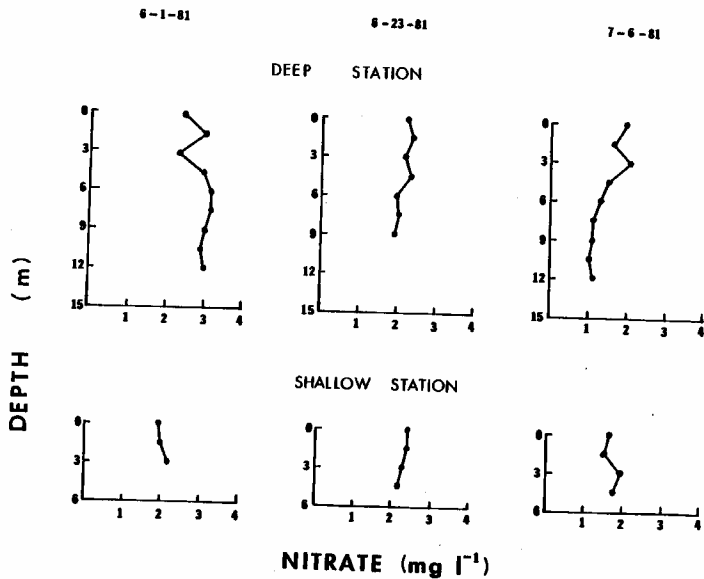




Fig. 11a-d. Temporal variation in nitrate profiles at the deep and shallow stations of Lake of the Woods.







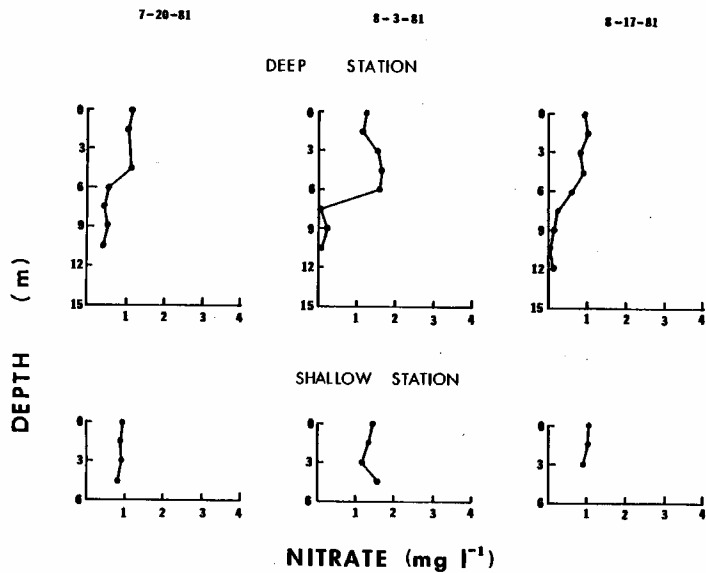




Fig. 12a-d. Temporal variation in organic nitrogen profiles at the deep and shallow stations of Lake of the Woods.

1-20-81

2-17-81

3-30-81

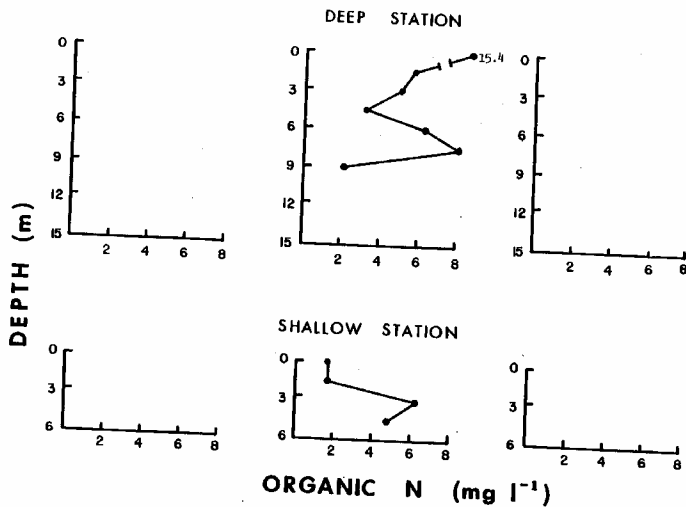
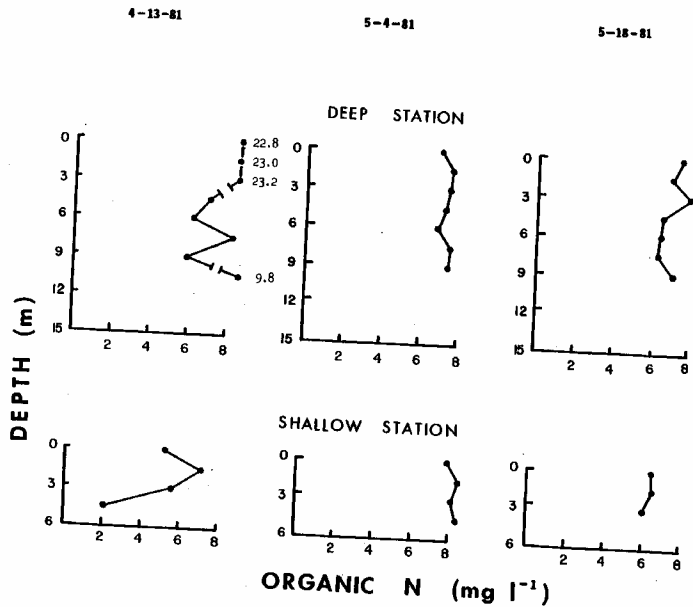
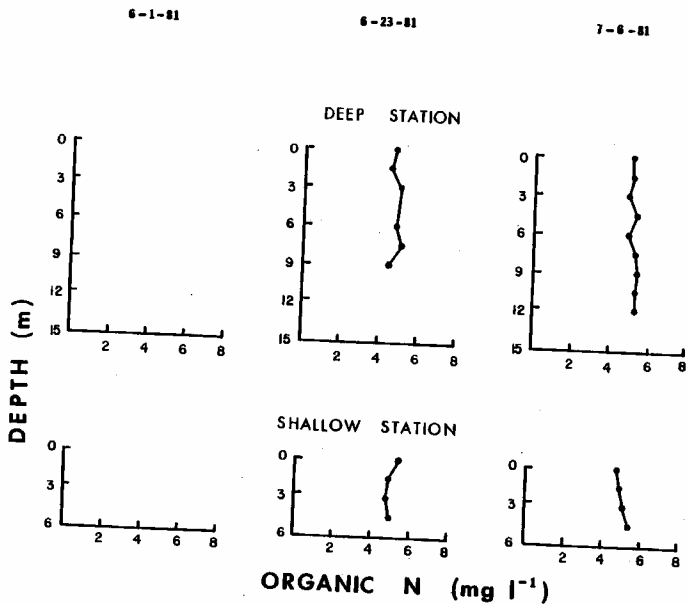


Fig. 12a





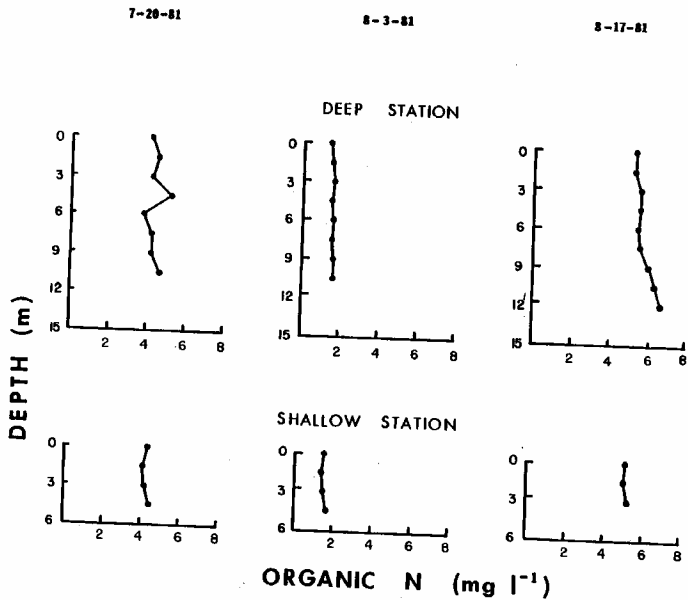
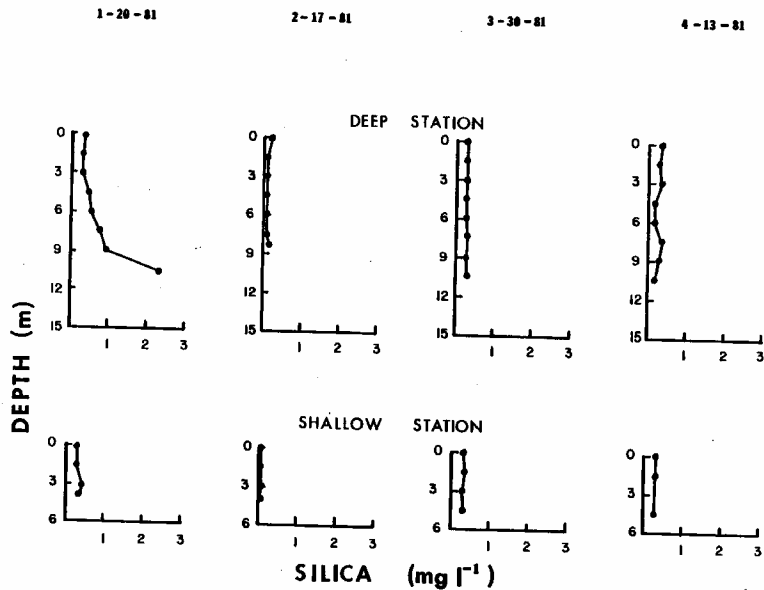
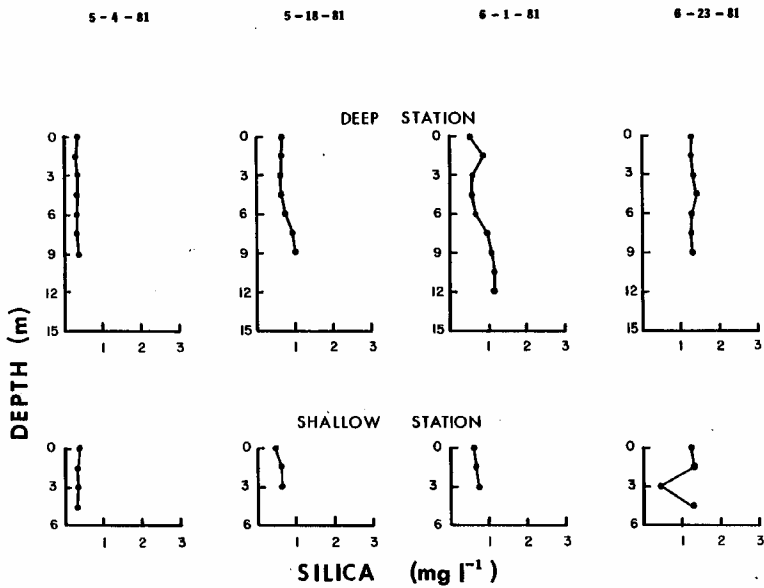




Fig. 13a-c. Temporal variation in silica  
profiles at the deep and shallow  
stations of Lake of the Woods.





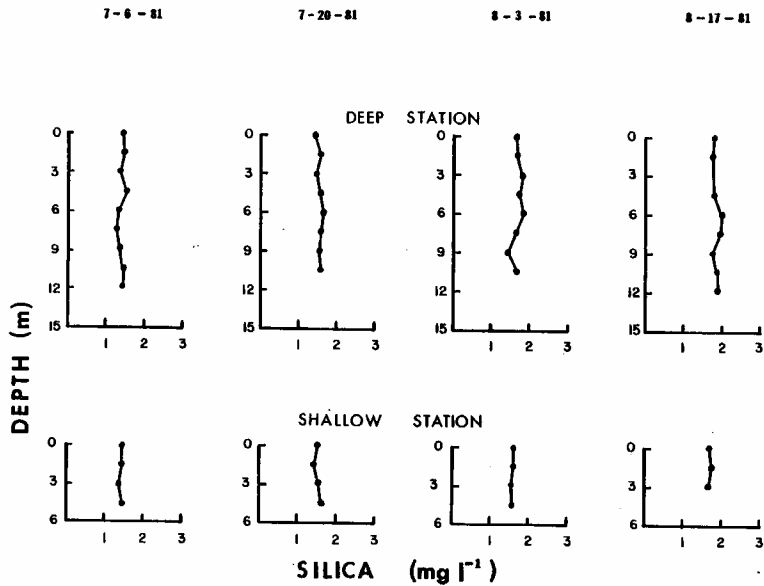
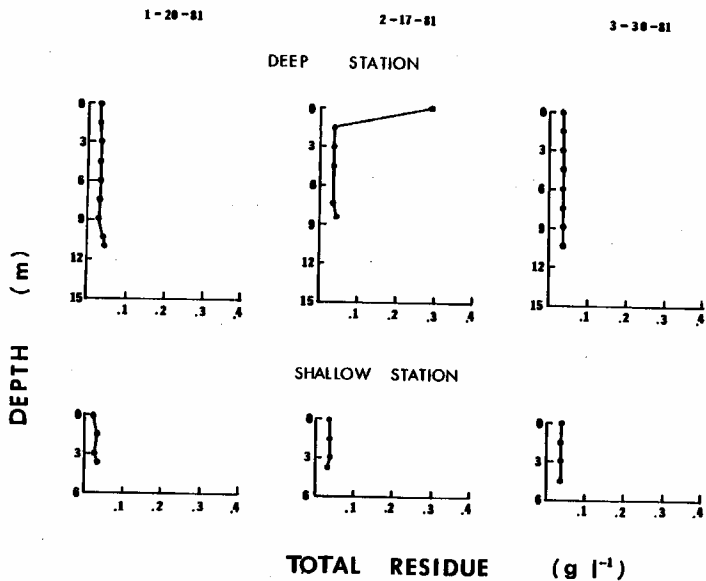
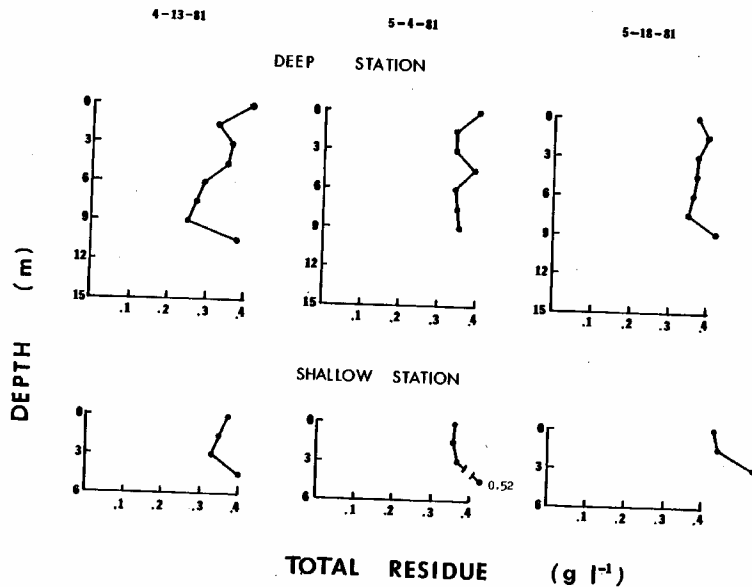
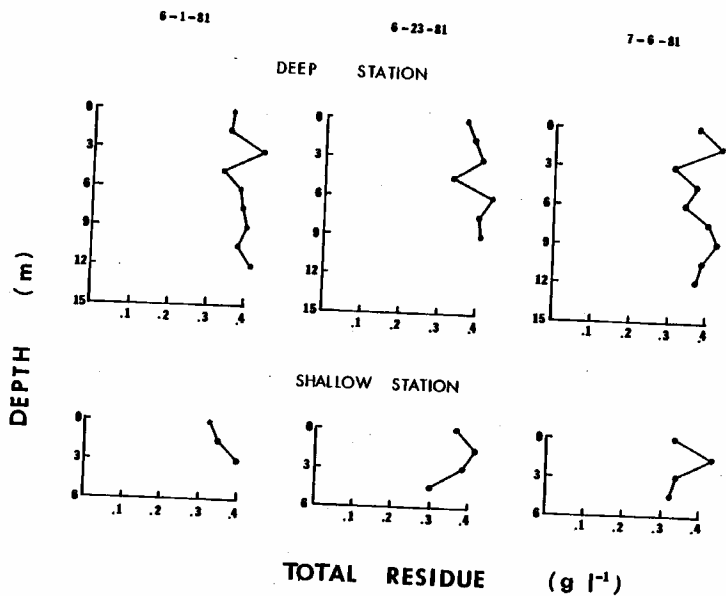


Fig. 13c

Fig. 14a-d. Temporal variation in total  
residue profiles at the deep  
and shallow stations of Lake of  
the Woods.







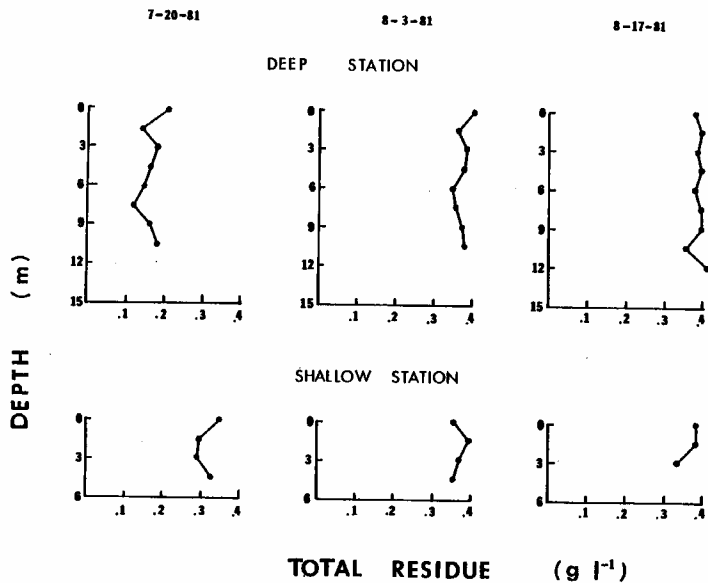
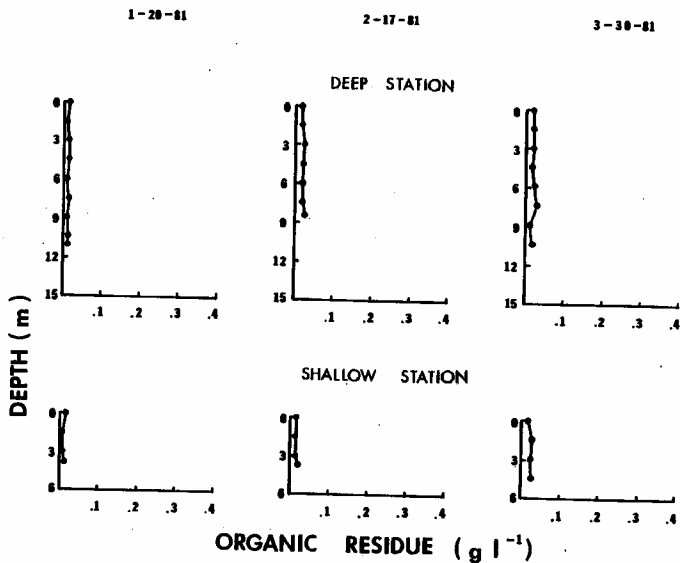
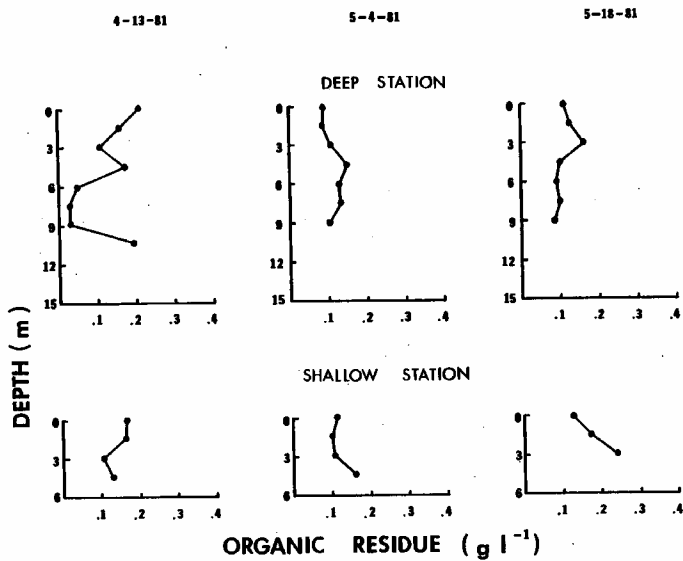
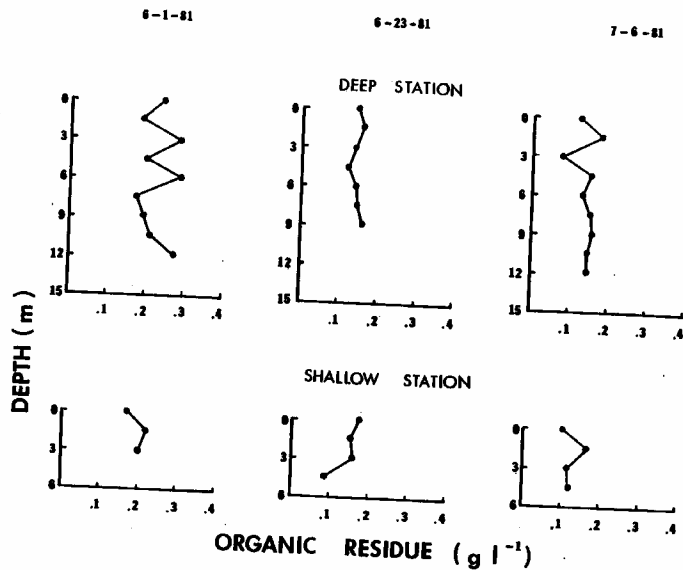




Fig. 15a-d. Temporal variation in organic residue profiles at the deep and shallow stations of Lake of the Woods.







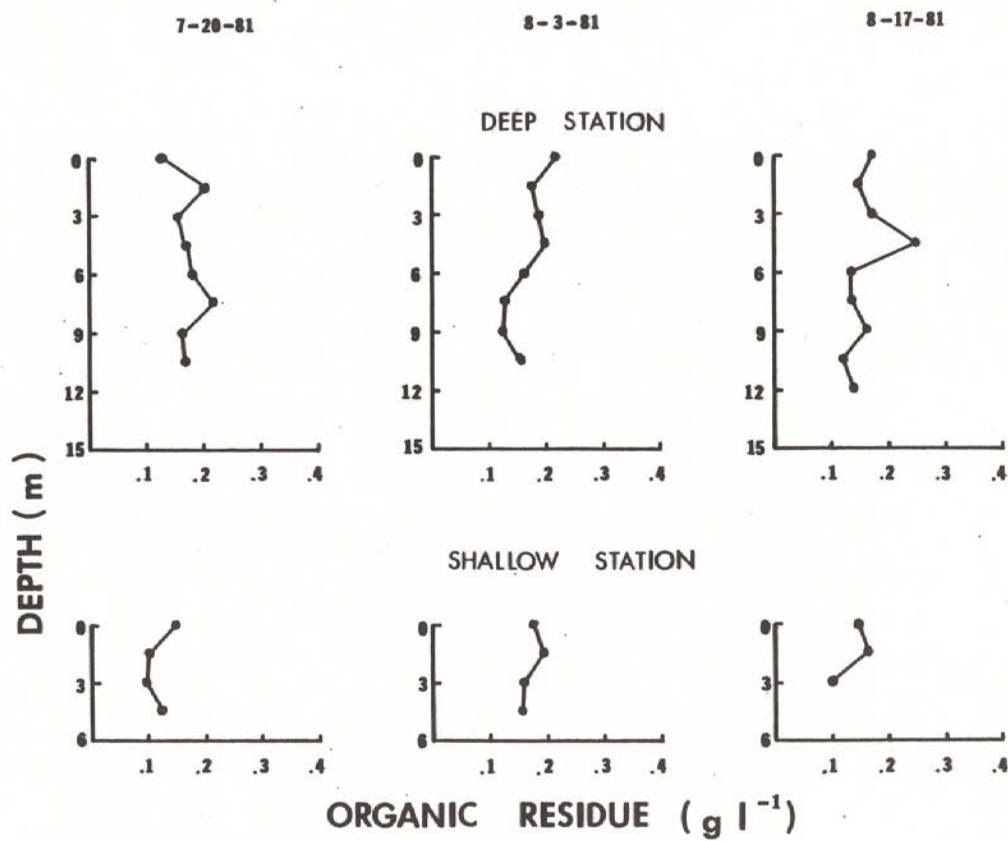
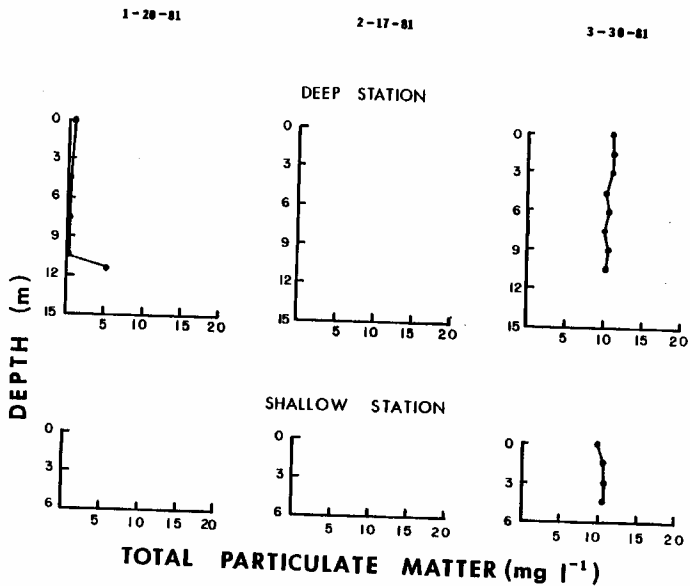
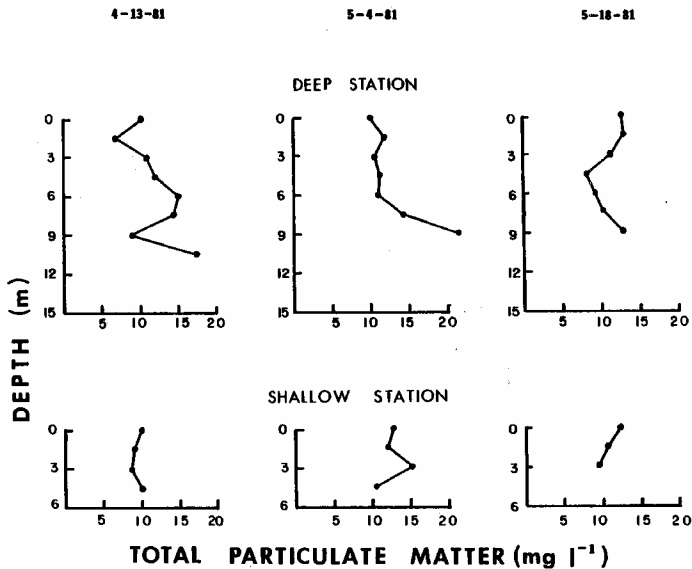




Fig. 16a-d. Temporal variation in total  
particulate matter profiles at  
the deep and shallow stations  
of Lake of the Woods.





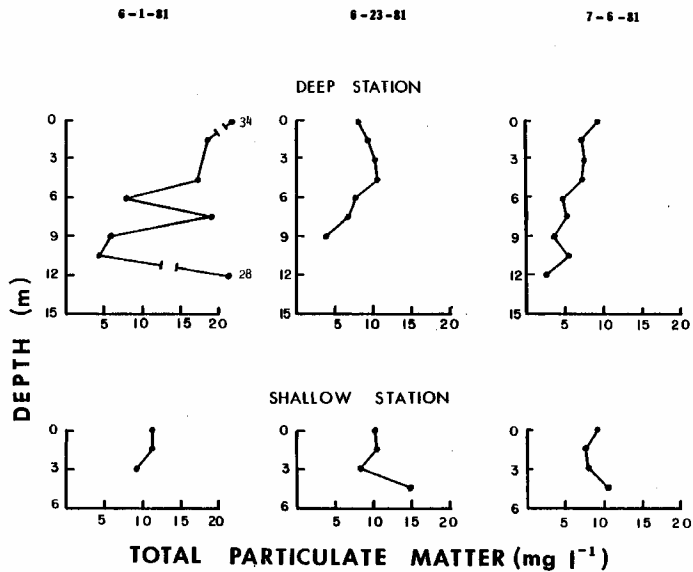


Fig. 16c

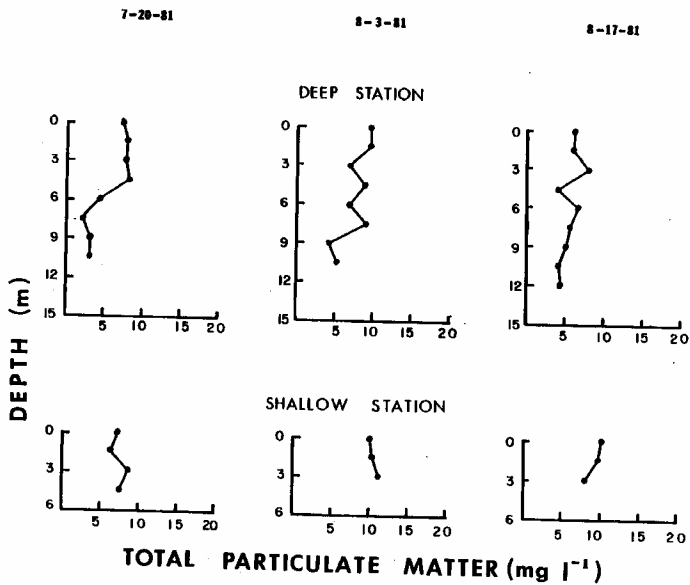




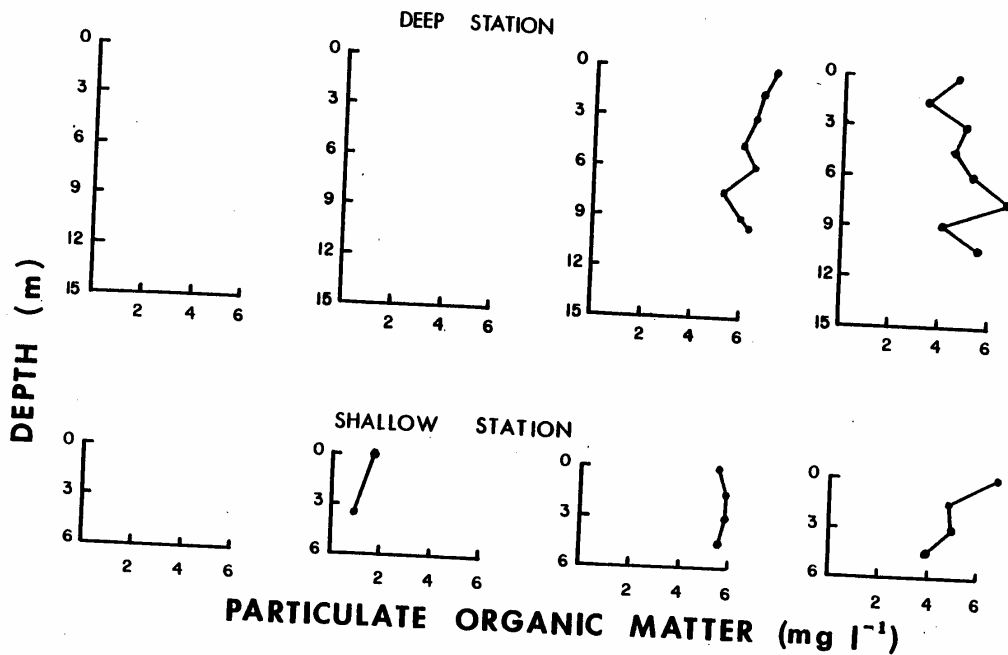
Fig. 17a-c. Temporal variation in particulate organic matter profiles at the deep and shallow stations of Lake of the Woods.

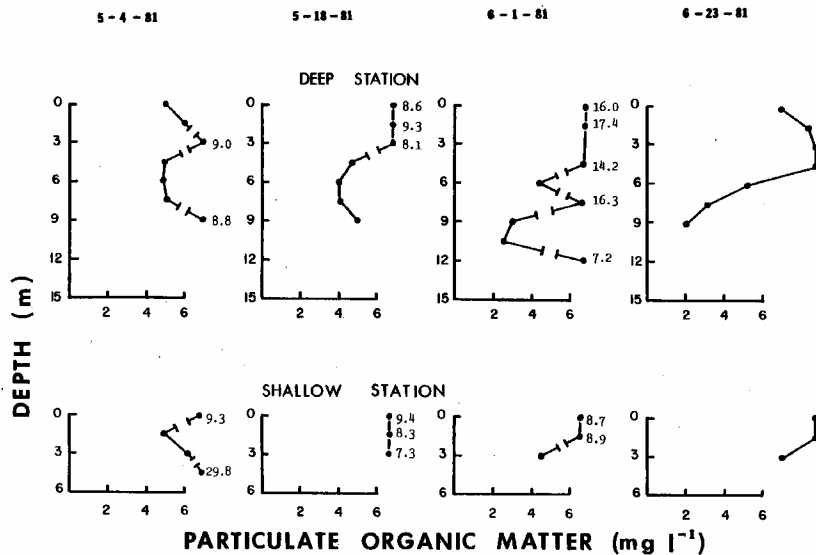
1-20-81

2-17-81

3-30-81

4-13-81





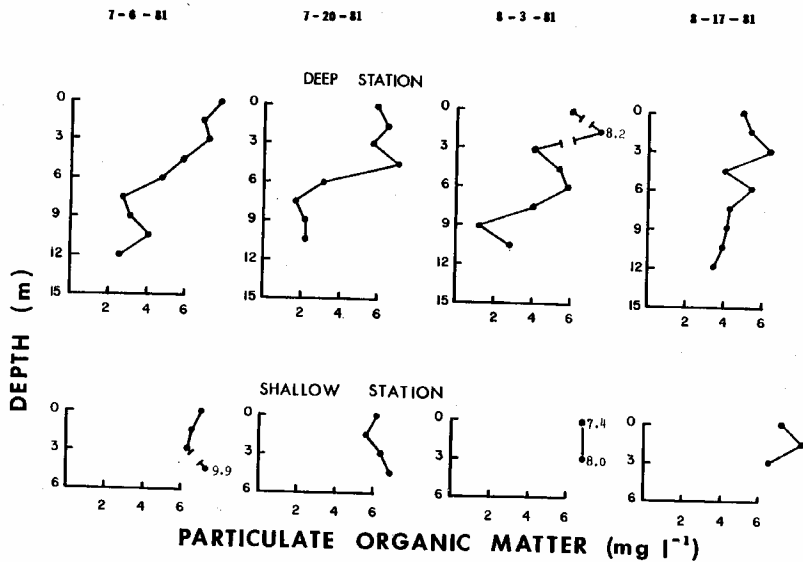


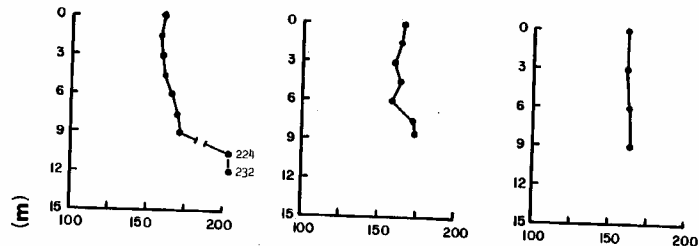
Fig. 18a-d. Temporal variation in total alkalinity profiles at the deep and shallow stations of Lake of the Woods.

1-20-81

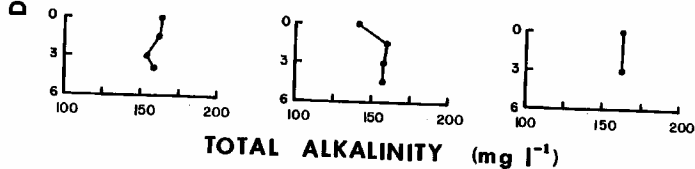
2-17-81

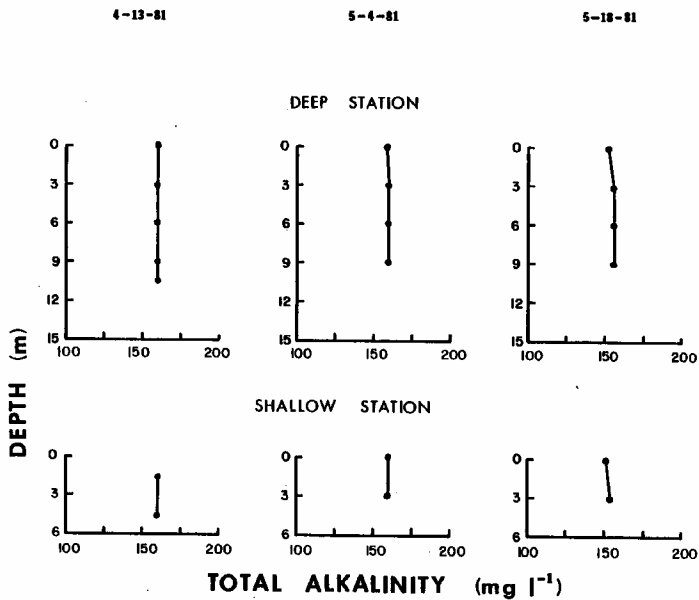
3-30-81

## DEEP STATION



## SHALLOW STATION



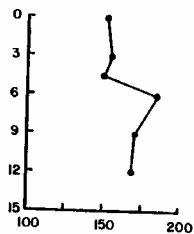
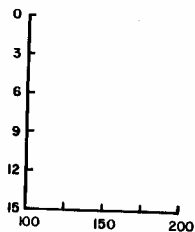
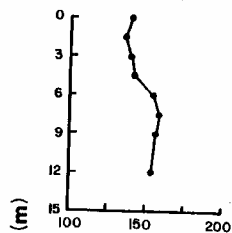


6-1-81

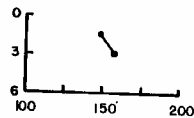
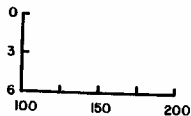
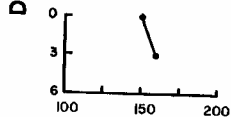
6-23-81

7-6-81

## DEEP STATION



## SHALLOW STATION

TOTAL ALKALINITY (mg l<sup>-1</sup>)

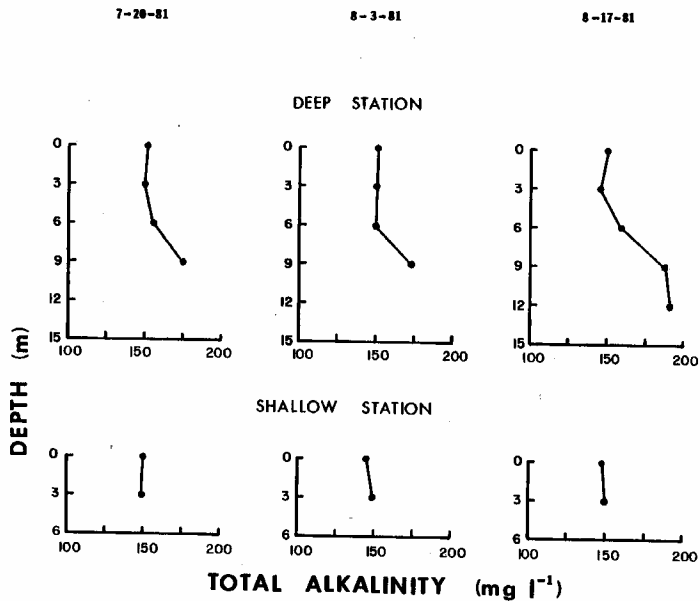


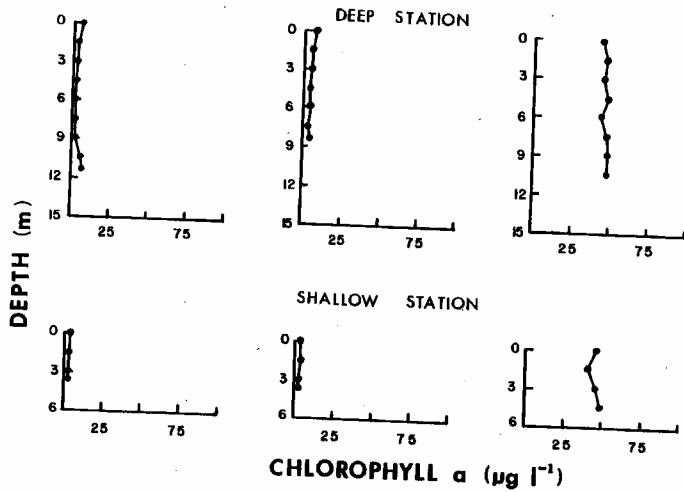


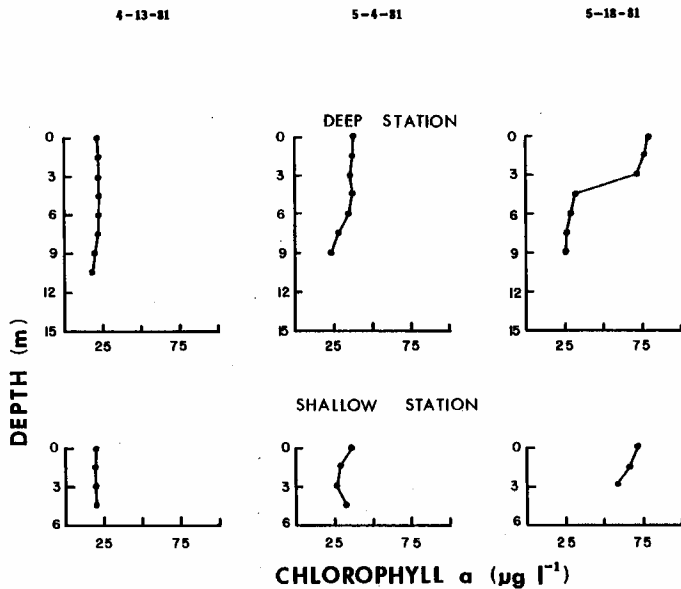
Fig. 19a-d. Temporal variation in chlorophyll a  
profiles at the deep and shallow  
stations of Lake of the Woods.

1-20-81

2-17-81

3-30-81

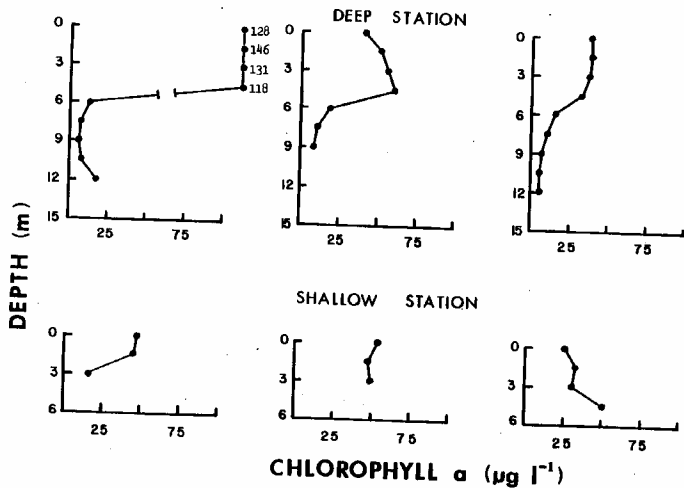




6-1-81

6-23-81

7-6-81



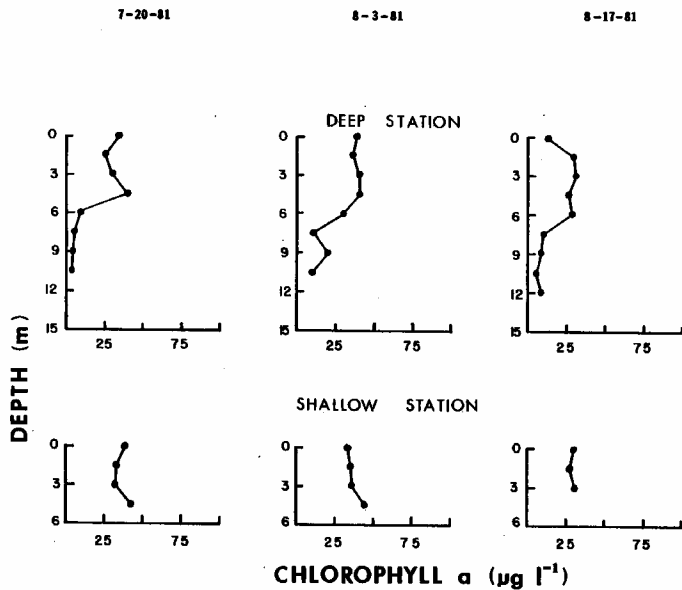
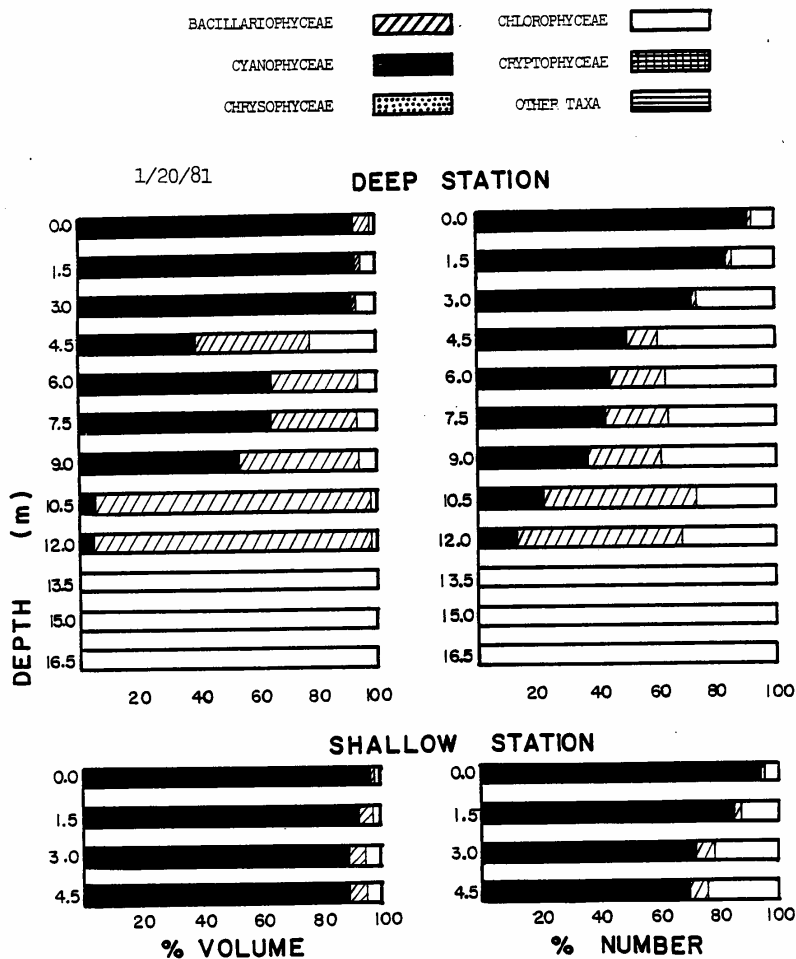
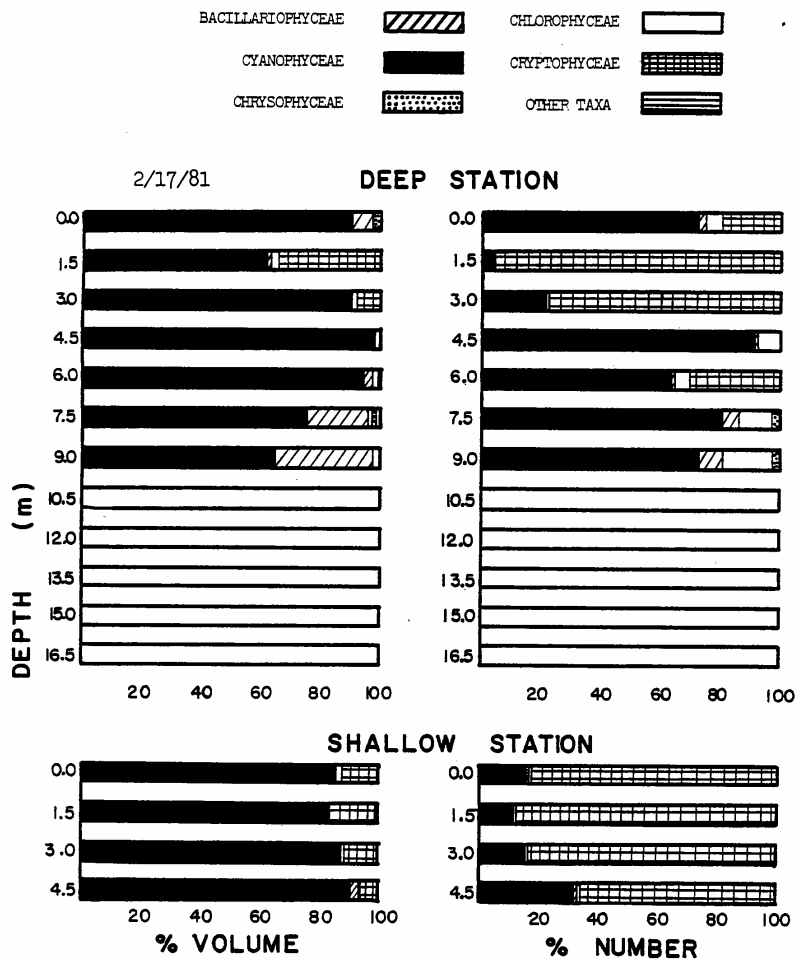
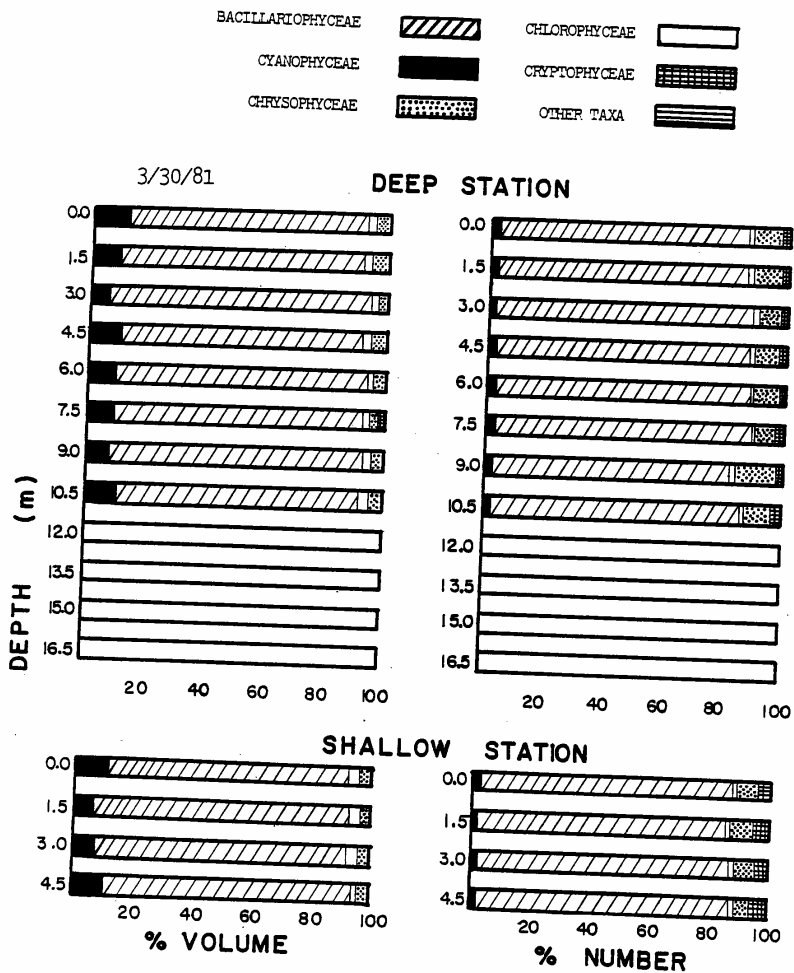


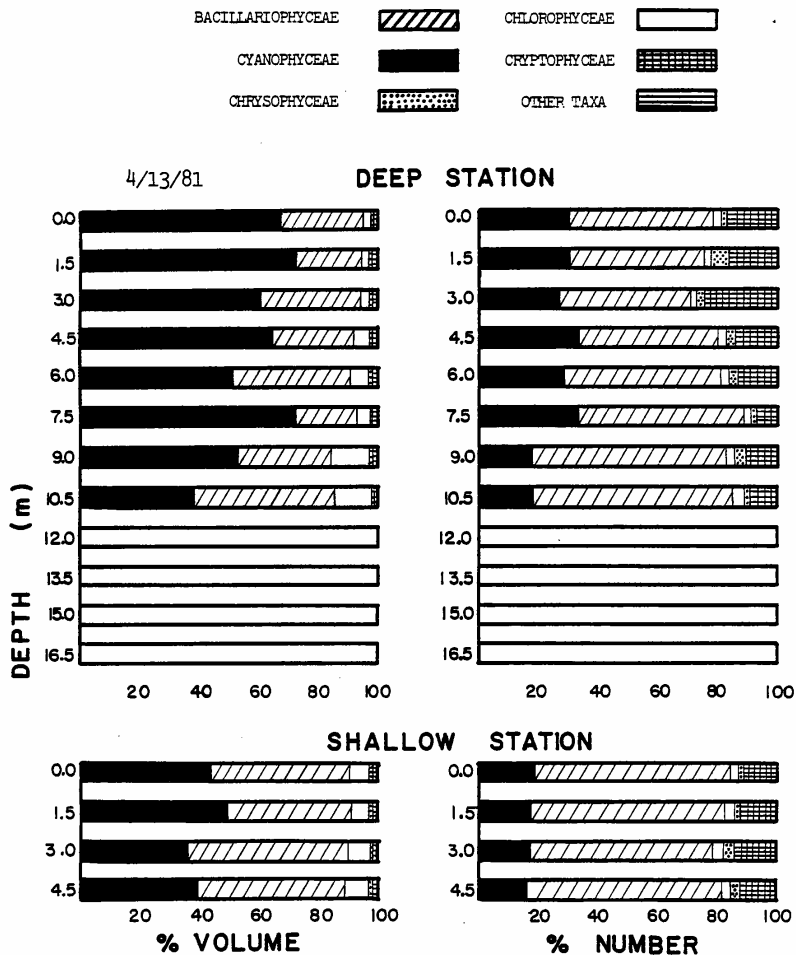


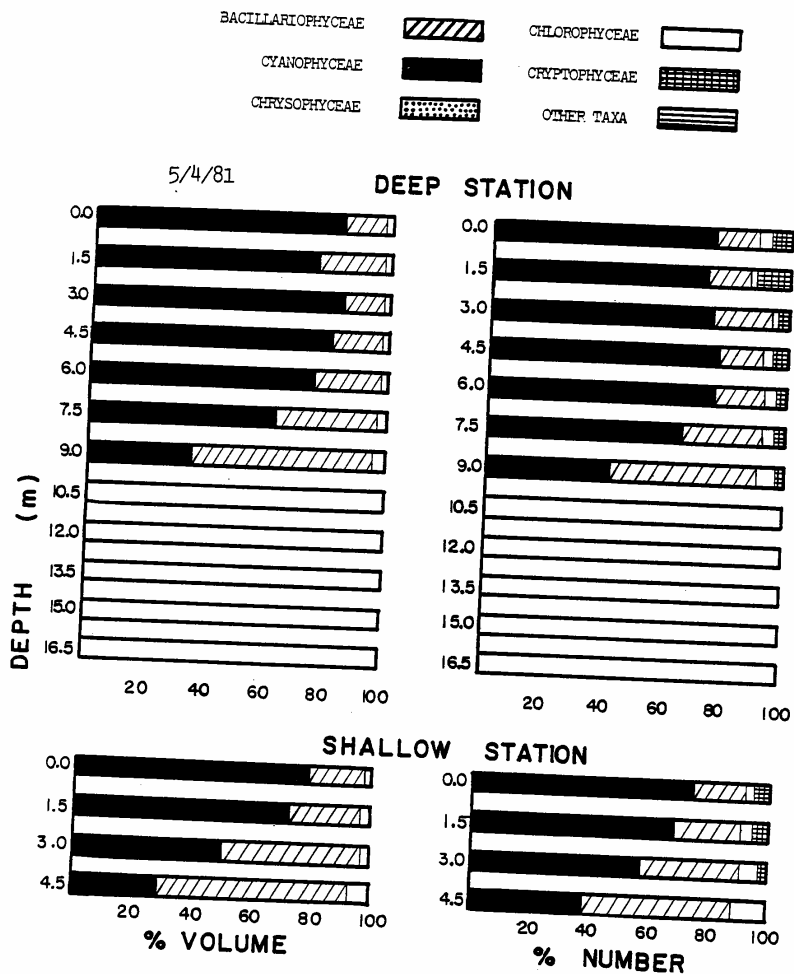
Fig. 20a-k. Phytoplankton community  
composition on various dates  
in Lake of the Woods.

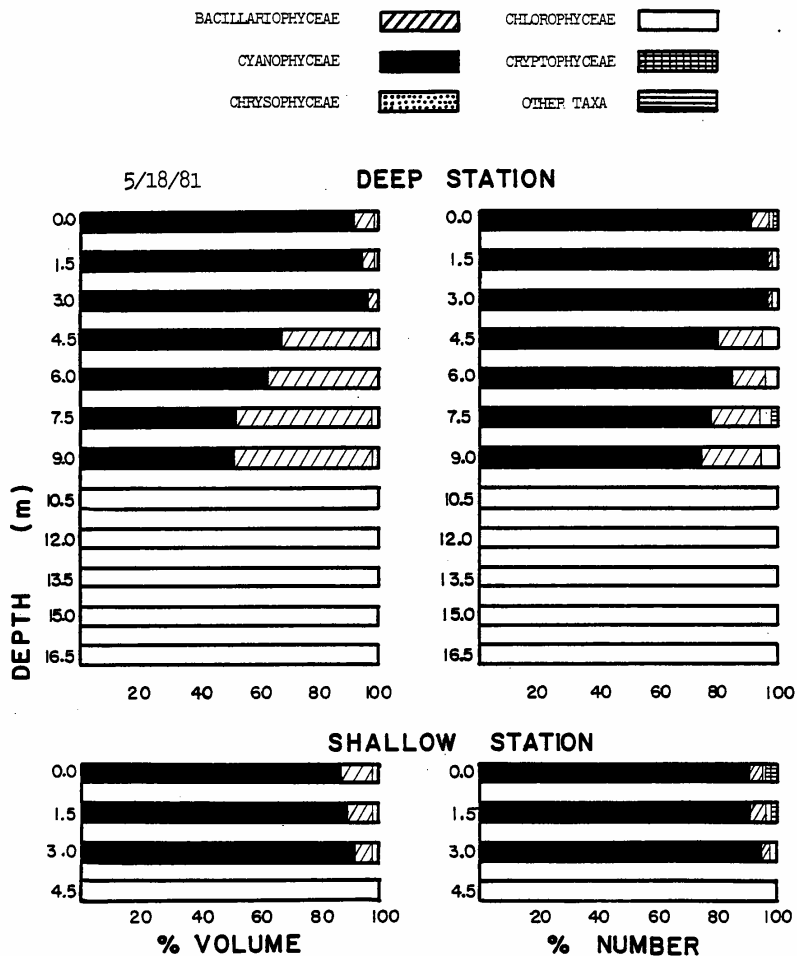


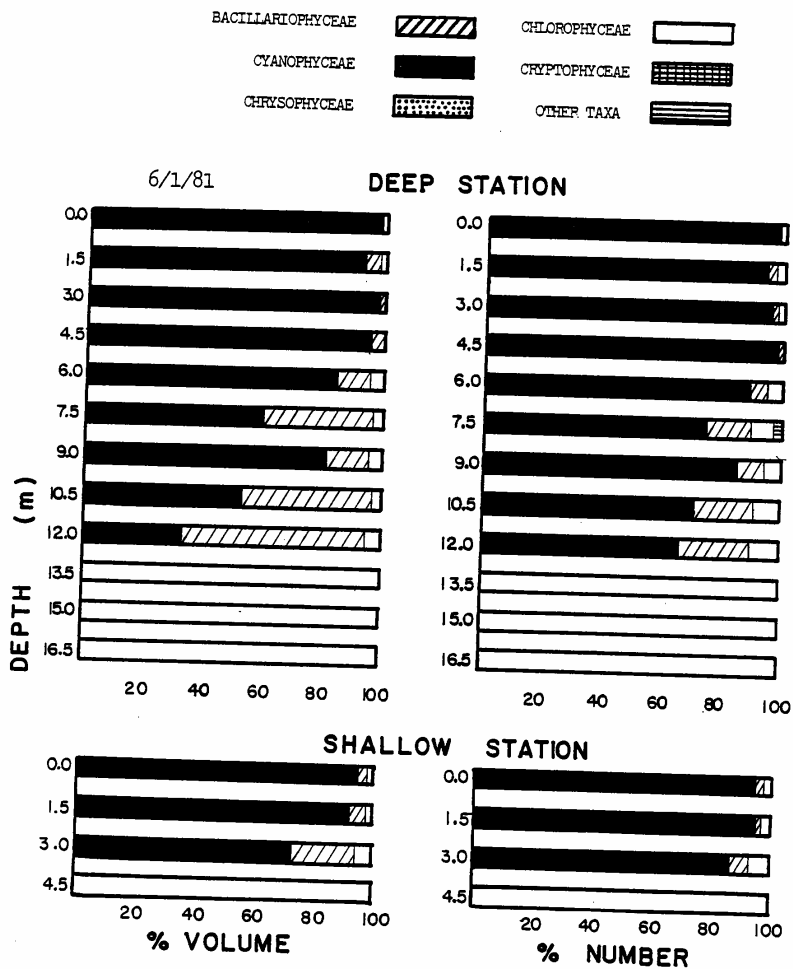


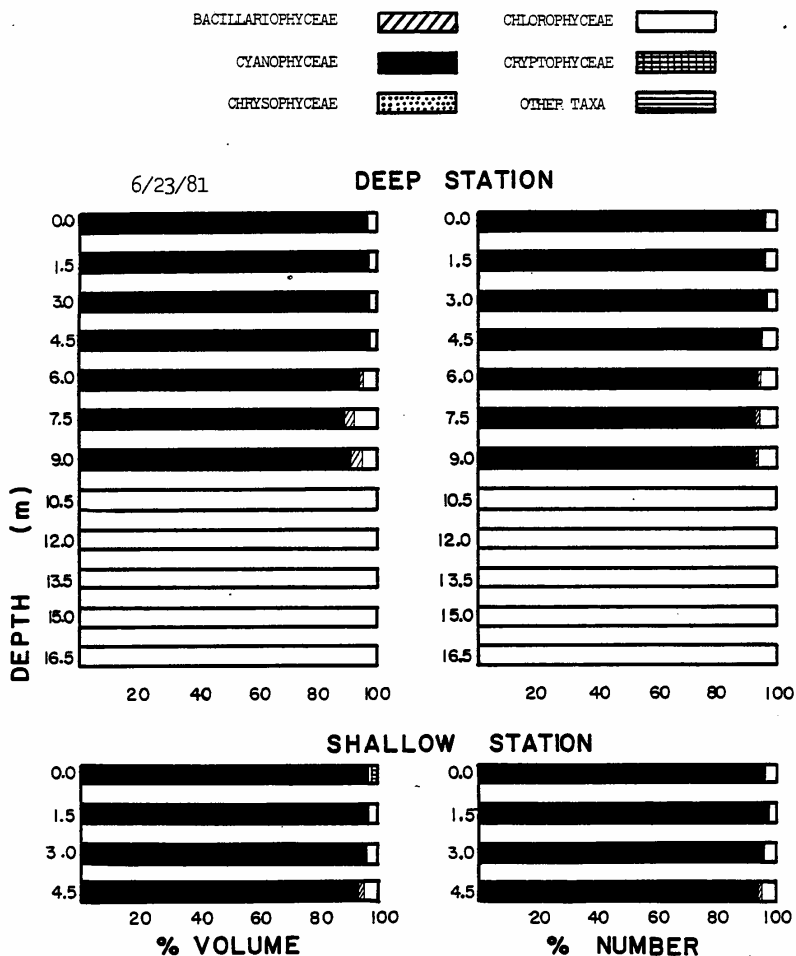


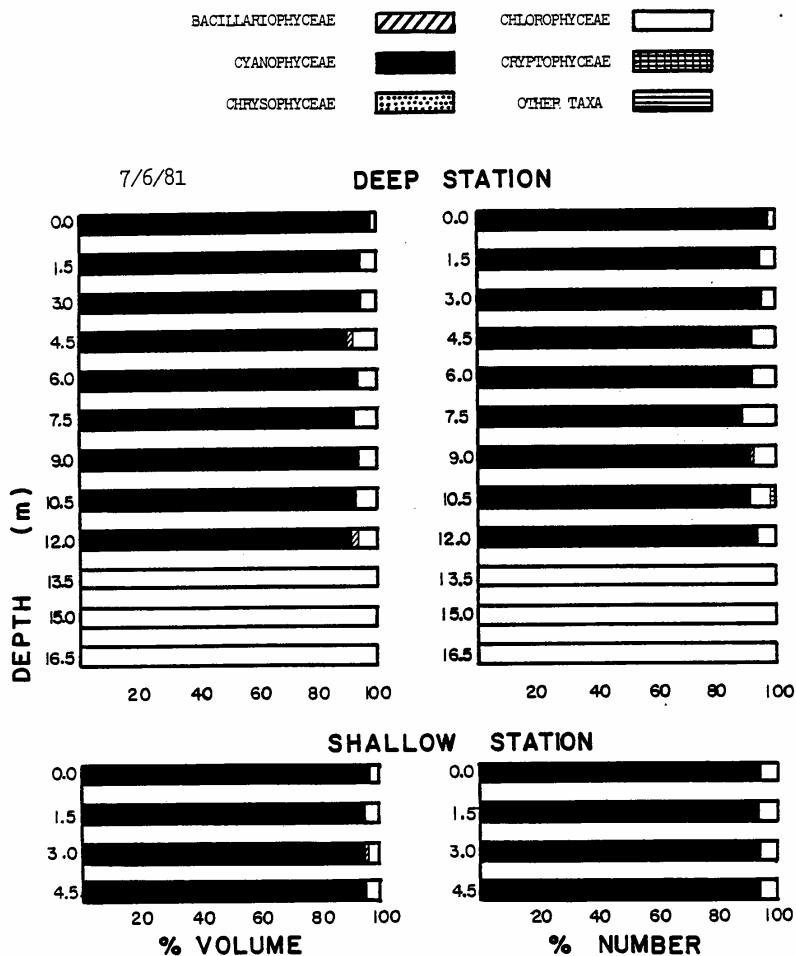


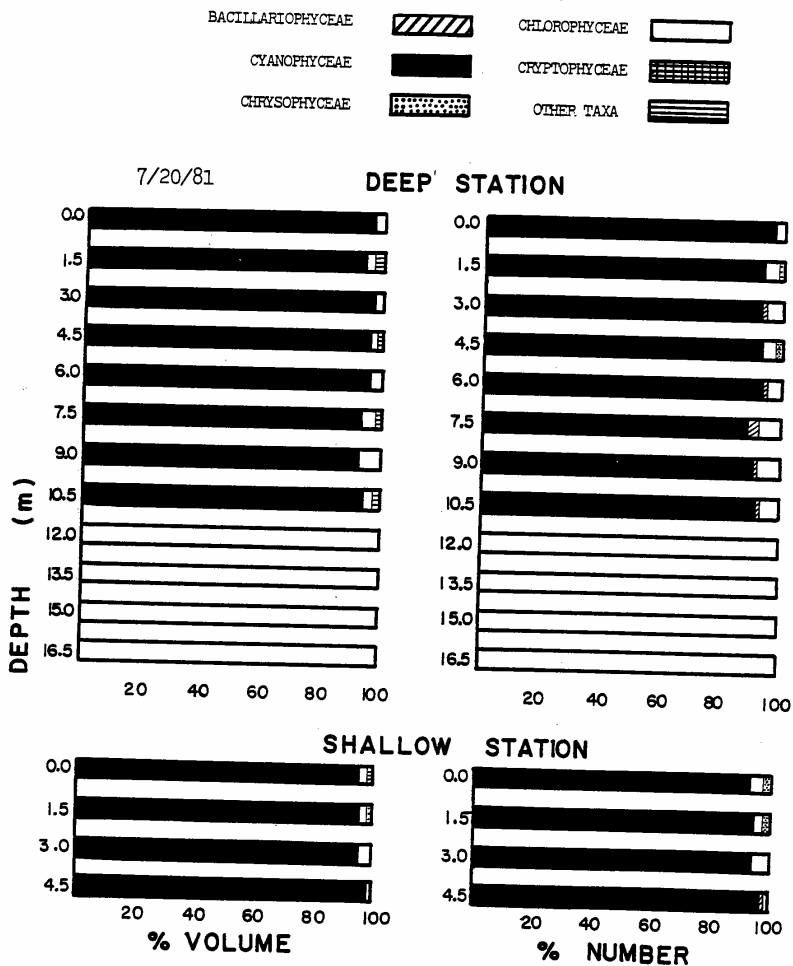












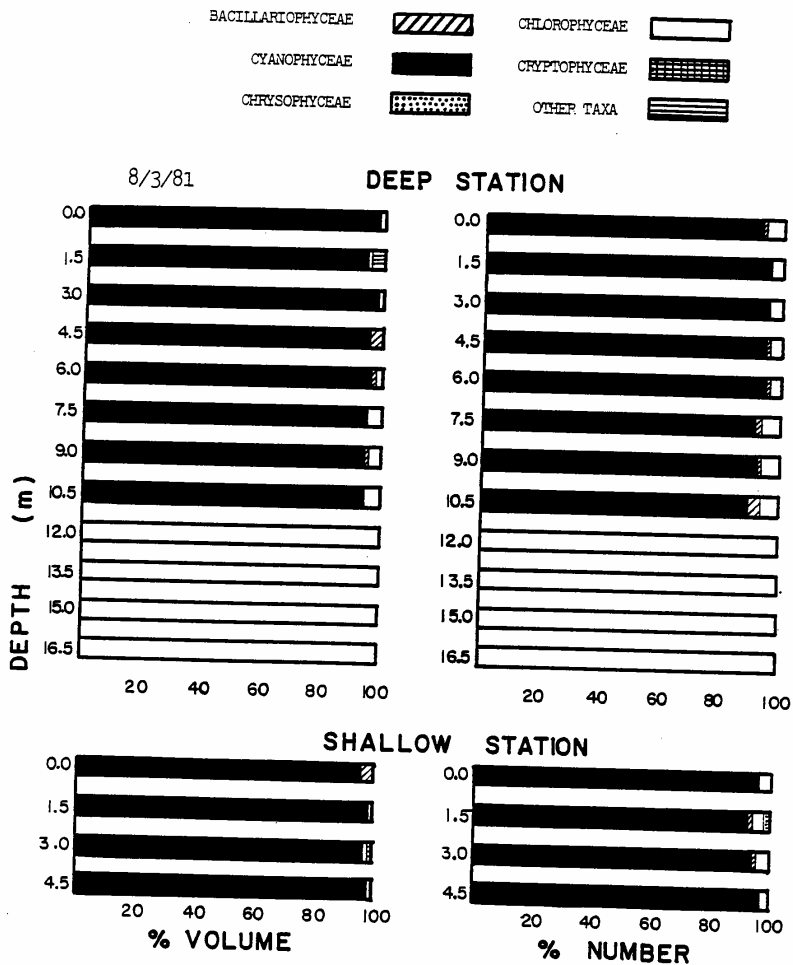


Fig. 21. Phytoplankton numbers and volume at the deep {0-4.5m ( $\Delta$ ); 6-10.5m (o)} and shallow ( $\bullet$ ) stations of Lake of the Woods.

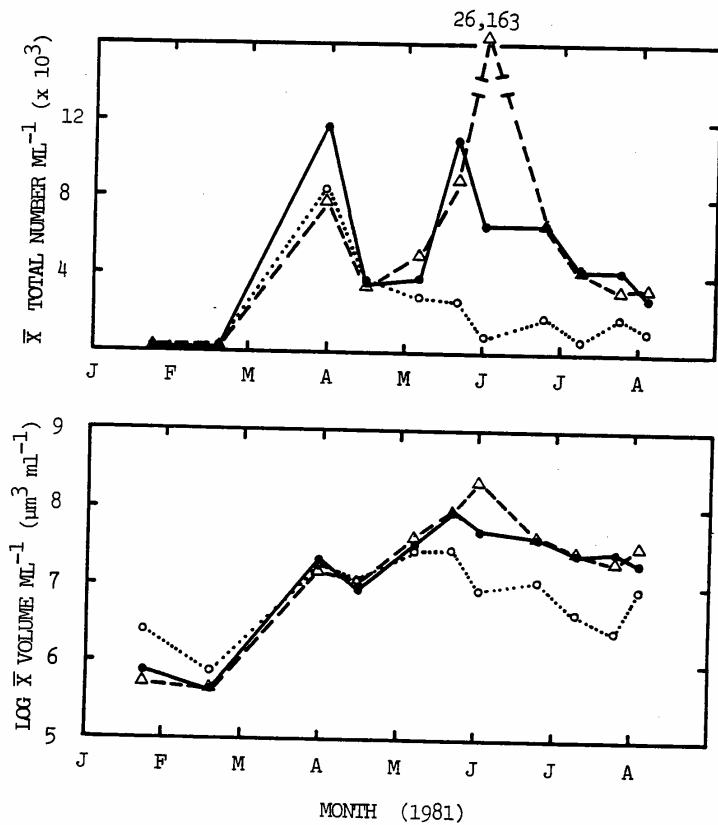


Fig. 22. Integral photosynthesis on  
three dates in Lake of the  
Woods.

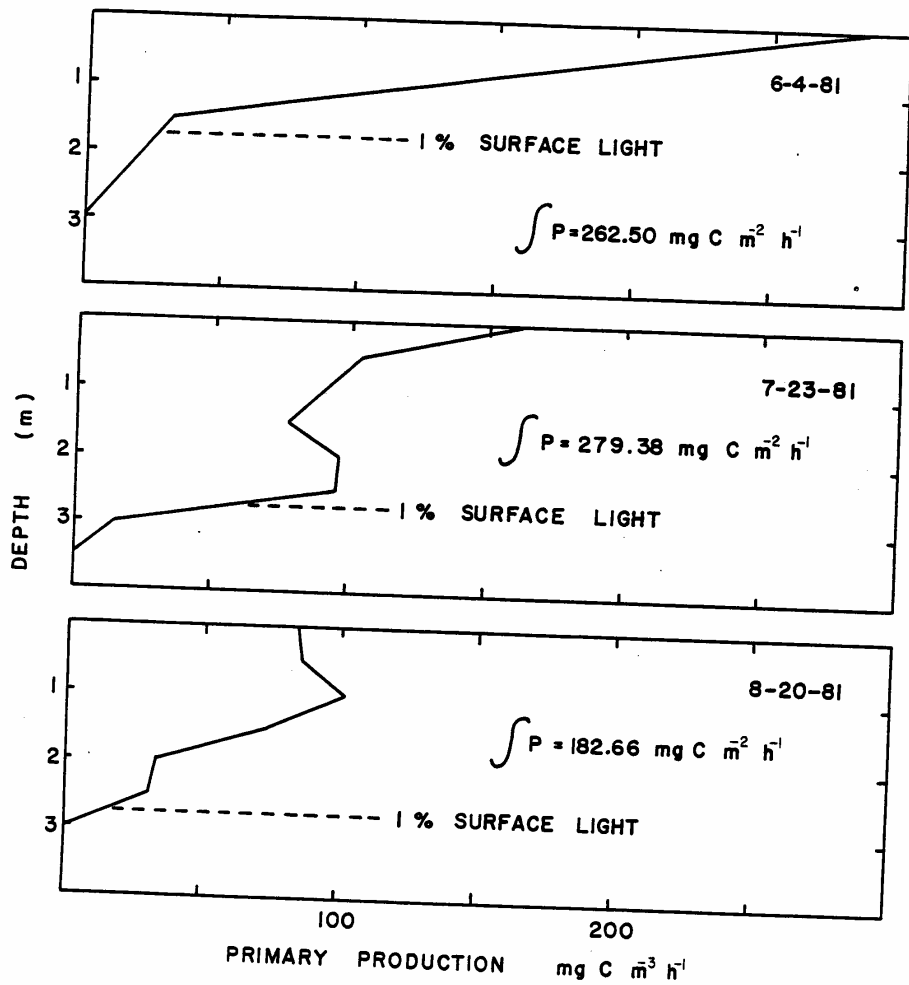


Fig. 23. Algal bioassay of Lake of the Woods on August 3, 1981. See text for explanation of symbols.

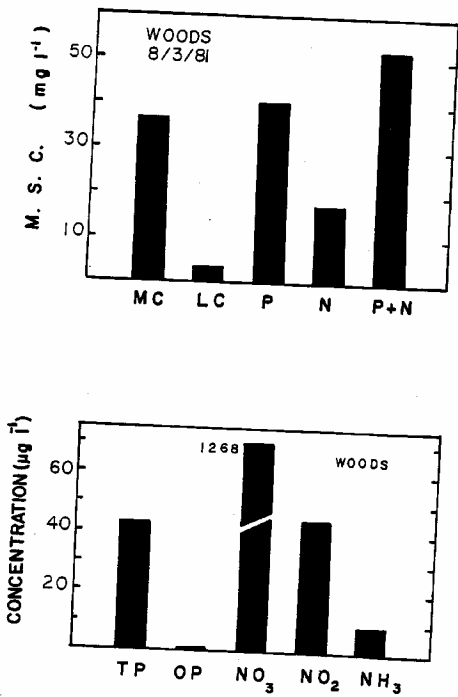


Fig. 24. Temporal variation in the macrophyte biomass present in Lake of the Woods during the 1981 growing season.

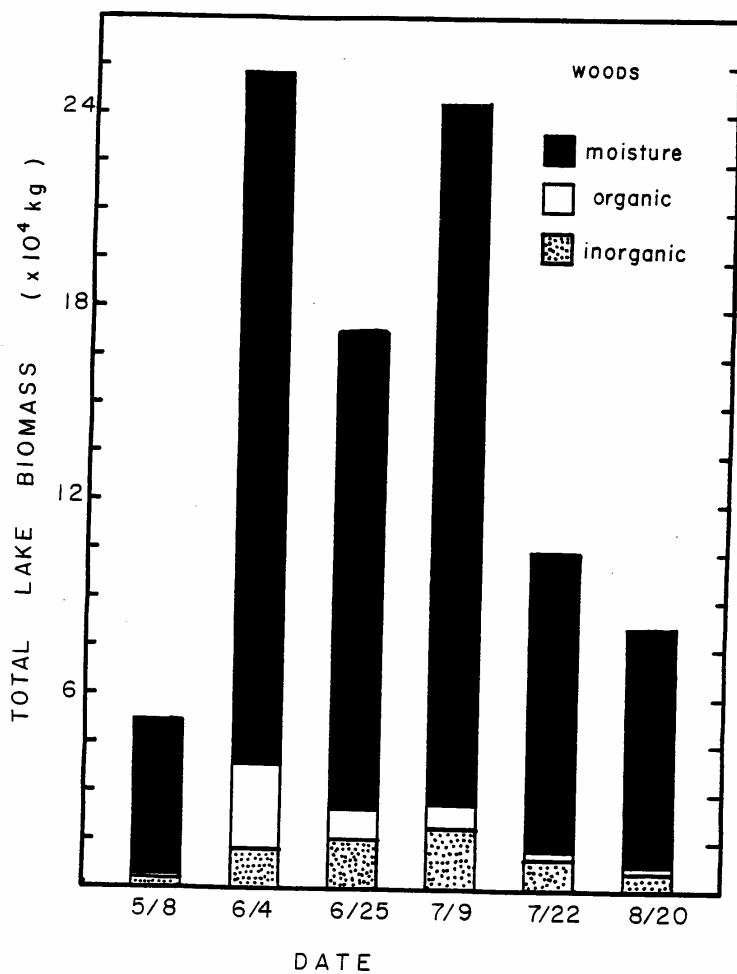


Fig. 25. Temporal variation in the nutrient pools present in the macrophyte biomass of Lake of the Woods during the 1981 growing season.

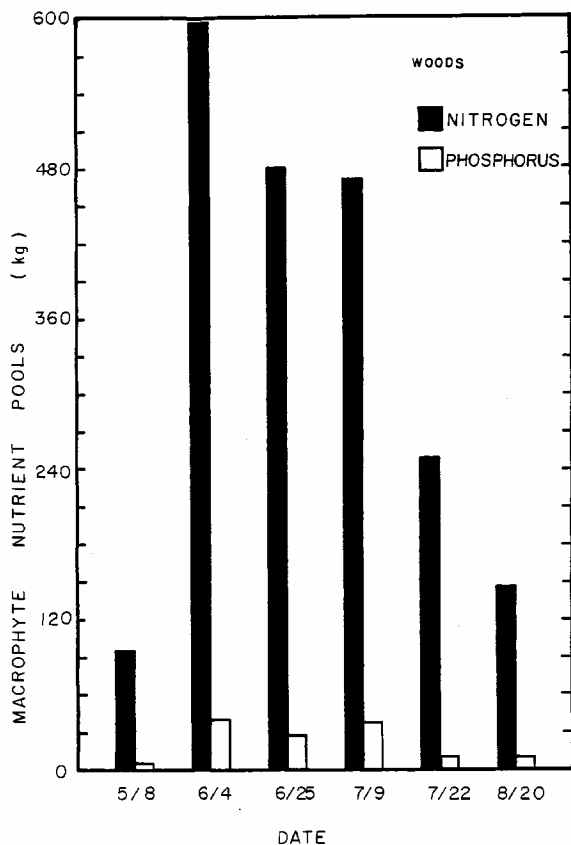


TABLE 1.

Total attenuation ( $n$ ) and non-chlorophyll attenuation ( $n_w$ ) coefficients of the upper 3 m of Lake of the Woods on various dates.

Date	$n$	$n_w$	$n_w/n$
6/1/81	2.38	0.22	0.09
6/4/81	3.36	1.20	0.36
6/23/81	1.98	1.19	0.60
7/6/81	1.58	0.96	0.61
7/23/81	1.39	0.89	0.64
8/17/81	1.56	1.15	0.74
8/20/81	1.38	0.96	0.70

TABLE 2

Total coliform counts for Lake of the Woods, the  
tributaries and the outlet.

Site	4-13	5-18	6-1	7-6	7-20*	8-3
1	410	ND**	3100	3200 (3200)	440 (440)	47000 (47000)
2	690	ND	700	4300 (4300)	400 (400)	1500 (1500)
3	5000	ND	6800	2500 (2500)	6000 (6000)	100 (100)
4	6300	ND	2000	6700 (6700)	5200 (5200)	4600 (4600)
5	6900	ND	150	5500 (5500)	1500 (1500)	2600 (2600)
6	830	ND	2800	1900 (1900)	1200 (1200)	1900 (1900)
7	3800	ND	150	70 (70)	1600 (1600)	40 (40)
Shallow	50	730	15	30 (30)	10 (10)	3 (3)
Deep	260	1000	15	20 (20)	40 (40)	75 (75)

\* Sample taken 1-3 m from shore

\*\* No Data

( ) Verified Values

TABLE 3

Fecal coliform counts for Lake of the Woods,  
the tributaries, and the outlet.

Site	Sample Dates (1981)					
	4-13	5-18	6-1	7-6	7-20*	8-3
1	3100	ND**	2400	1400 (990)	700 (630)	2500 (1300)
2	110	ND	510	1300 (880)	430 (390)	900 (450)
3	4000	ND	4000	3200 (2200)	230 (250)	420 (220)
4	3000	ND	2400	2000 (1400)	6000 (5400)	2800 (1400)
5	0	ND	400	1700 (1200)	660 (590)	740 (370)
6	180	ND	1600	1120 (780)	740 (660)	280 (140)
7	130	ND	250	340 (280)	880 (790)	30 (15)
Shallow	10	10	50	15 (10)	15 (13)	<3 (<2)
Deep	20	10	10	10 (7)	10 (9)	10 (5)

\* Sample taken 1-3 m from shore

\*\* No Data

( ) Verified Values

TABLE 4

Fecal streptococci counts for Lake of the Woods,  
the tributaries, and the outlet.

Site	4-13	5-18	6-1	7-6	7-20*	8-3
1	350	ND**	2400	2800 (2800)	4900 (4900)	2100 (2100)
2	770	ND	810	2900 (2900)	1700 (1700)	1400 (1400)
3	110	ND	28000	1400 (1400)	16200 (16200)	1600 (1600)
4	1400	ND	2000	6700 (6700)	54800 (54800)	4200 (4200)
5	390	ND	2500	3300 (3300)	4000 (4000)	3900 (3900)
6	60	ND	2700	6200 (6200)	1400 (1400)	1200 (1200)
7	1000	ND	390	230 (230)	2400 (2400)	200 (200)
Shallow	0	120	20	30 (30)	480 (480)	60 (60)
Deep	70	140	140	430 (430)	860 (860)	280 (280)

\* Sample taken 1-3 m from shore

\*\* No Data

( ) Verified Values

TABLE 5

Fecal coliform: fecal streptococci ratios for Lake of the Woods  
bacteriological data.

Site	Sample Dates (1981)					
	4-13	5-18	6-1	7-6	7-20*	8-3
1	8.86	ND**	1.00	0.51	0.14	1.22
2	0.14	ND	0.64	0.43	0.26	0.63
3	3.74	ND	0.14	3.65	0.01	0.27
4	2.16	ND	1.20	0.30	0.11	0.67
5	0.0	ND	0.16	0.52	0.17	0.19
6	-	ND	0.59	0.18	0.53	0.24
7	0.13	ND	0.69	1.50	0.37	-
Shallow	-	.08	-	-	0.03	-
Deep	-	0.13	-	0.03	0.00	0.04

\* Sample taken 1 m from shore

\*\* No Data

TABLE 6

Standard plate counts (2 day) for Lake of the Woods,  
the tributaries, and the outlet.

Site	Date (1981)					
	4-13	5-18	6-1	7-6	7-20*	8-3
1	ND**	ND	$2.4 \times 10^3$	$2.0 \times 10^3$	$2.2 \times 10^4$	$2.0 \times 10^4$
2	ND	ND	$1.0 \times 10^3$	$3.7 \times 10^3$	$4.1 \times 10^3$	$1.1 \times 10^4$
3	$5.5 \times 10^3$	ND	$2.3 \times 10^4$	$1.5 \times 10^4$	$1.7 \times 10^4$	-
4	$4.0 \times 10^3$	ND	$9.9 \times 10^3$	$1.0 \times 10^5$	$2.2 \times 10^5$	$1.5 \times 10^5$
5	-	ND	$5.0 \times 10^3$	$1.2 \times 10^4$	$9.5 \times 10^3$	$2.2 \times 10^2$
6	ND	ND	$1.8 \times 10^4$	$7.4 \times 10^4$	$1.7 \times 10^5$	$3.0 \times 10^5$
7	-	ND	$9.1 \times 10^4$	$1.3 \times 10^4$	$5.3 \times 10^3$	$6.2 \times 10^4$
Shallow	-	$1.9 \times 10^2$	$1.8 \times 10^4$	$9.6 \times 10^3$	$9.1 \times 10^2$	$2.4 \times 10^3$
Deep	-	$3.7 \times 10^2$	$1.7 \times 10^2$	$5.5 \times 10^3$	$4.1 \times 10^3$	$3.1 \times 10^3$

\* Sample taken 1-3 m from shore

\*\* No Data

TABLE 7

List of macrophyte species present  
in Lake of the Woods.

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Myriophyllum spicatum

Potamogeton crispus

Potamogeton sp.

Nuphar advena

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TABLE 8.

## Zooplankton species found in Lake of the Woods

	Winter	Spring	Summer
<b>Rotifera</b>			
<i>Kellicottia bostoniensis</i> (Rousselet 1908)			X
<i>Kellicottia longispina</i> (Kellicott 1879)		X	X
<i>Keratella cochlearis</i> (Gosse 1851)	X	X	X
<i>Keratella quadrata</i> (Müller 1786)	X	X	X
<i>Notholca acuminata</i> var. <i>extensa</i> Olofsson 1918	X		
<i>Notholca michiganensis</i> Stemberger 1976	X		
<i>Trichocerca multicarinis</i> (Kellicott 1879)	X		
<i>Asplanchna priodonta</i> Gosse 1850			X
<i>Polyarthra dolichoptera</i> Idelson 1925		X	X
<i>Syncheata</i> sp.	X	X	X
<i>Filinia terminalis</i> (Plate 1886)	X		
<i>Pompholyx sulcata</i> Hudson 1885		X	X
<i>Conochilus unicornis</i> Rousselet 1892			X
			X
<b>Cladocera</b>			
<i>Leptodora kindtii</i> (Focke 1844)			
<i>Diaphanosoma birgei</i> Korinek 1981			X
<i>Daphnia galeata mendotae</i> Birge 1918			X
<i>Daphnia retrocurva</i> Forbes 1882	X	X	X
<i>Bosmina longirostris</i> (O. F. Müller 1785)			X
<i>Eubosmina coregoni</i> (Baird 1857)	X	X	X
<i>Chydorus sphaericus</i> (O. F. Müller 1785)	X	X	X
			X
<b>Copepoda</b>			
<i>Skistodiaptomus oregonensis</i> (Lilljeborg 1889)	X	X	X
<i>Tropocyclops parrisi</i> (Fisher 1894)	X	X	
<i>Diaicyclops thomasi</i> (Forbes 1882)	X	X	
<i>Mesocyclops edax</i> (Forbes 1891)		X	
<i>Eucyclops serrulatus</i> (Fisher 1851) = <i>E. agilis</i> (Koch 1838)			X
			X

TABLE 9

Characteristics of sediment types collected in Lake of the Woods on 22 July, 1981. All data are mean values of several determinations. Coefficients of variation on each parameter ranged from 2.79 to 10.5%.

	DW ( $\text{g cm}^{-3}$ )	Water ( $\text{g cm}^{-3}$ )	% Organic
Type 1 (fine organic silt)	.1431	.9060	18.32
Type 2 (small gravel/sand)	1.2604	.4623	1.87

TABLE 10

Total phosphorus concentration in the sediment  
interstitial water collected at various locations  
in Lake of the Woods on 22 July, 1981.

Sample	Depth (m)	$\mu\text{g P l}^{-1}$
Transect A		
A	1.0	501.2
N	4.0	1058.4*
D	8.0	1730.3*
O	13.0	2044.3*
E	12.0	3363.4*
X	4.0	493.4*
H	3.0	542.1
Transect B		
L	4.5	540.4
S	0.5	190.5
Others		
F	3.0	1142.5
G	1.0	235.8
Y	12.0	1934.4
K	6.25	1153.0
I	1.0	216.6
R	4.0	510.8
B	0.5	141.8

\* Mean of three replicate determinations (coefficient of variation ranged from 0.62% to 5.22%).

TABLE 11

Concentration of acid-nonlabile sediment bound phosphorus collected at various locations in Lake of the Woods on 22 July, 1981.

Sample	Depth (m)	µg P/g Dry Wt. Sed.
Transect A		
A	1.0	609.1*
N	4.0	492.6*
D	8.0	695.6*
O	13.0	1053.4*
E	12.0	966.1*
X	4.0	94.0*
H	3.0	259.0*
Transect B		
L	4.5	547.0
S	0.5	76.6
Others		
F	3.0	465.2
G	1.0	198.1
Y	12.0	806.3
K	6.25	757.9
I	1.0	90.9
R	4.0	171.2
B	0.5	95.8

\* Mean of three replicate determinations (coefficient of variation ranged from 0.63% to 63.29%).

TABLE 12

Total sediment phosphorus pool of each  
depth interval in Lake of the Woods.

Stratum (ft.)	Total P g (sediment unit) <sup>-1</sup>	Area (m <sup>2</sup> x10 <sup>5</sup> )	P Pool (gx10 <sup>6</sup> )
0-5	5.790	5.2639	3.0478
6-10	5.258	3.1150	1.6379
11-15	7.347	2.1884	1.6078
16-20	9.149	2.0900	1.9121
21-25	10.950	1.8138	1.9861
26-30	10.111	1.6166	1.6345
31-35	11.517	1.5772	1.8165
36-40	12.922	1.1040	1.4266
41-45	15.259	0.7886	1.2033
46-48	15.259	0.0789	0.1203
TOTAL		19.7150	16.3929



## APPENDIX IV

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Fig. 1. Discharge curve for tributary  
sampled at Site 1, Lake of the  
Woods.

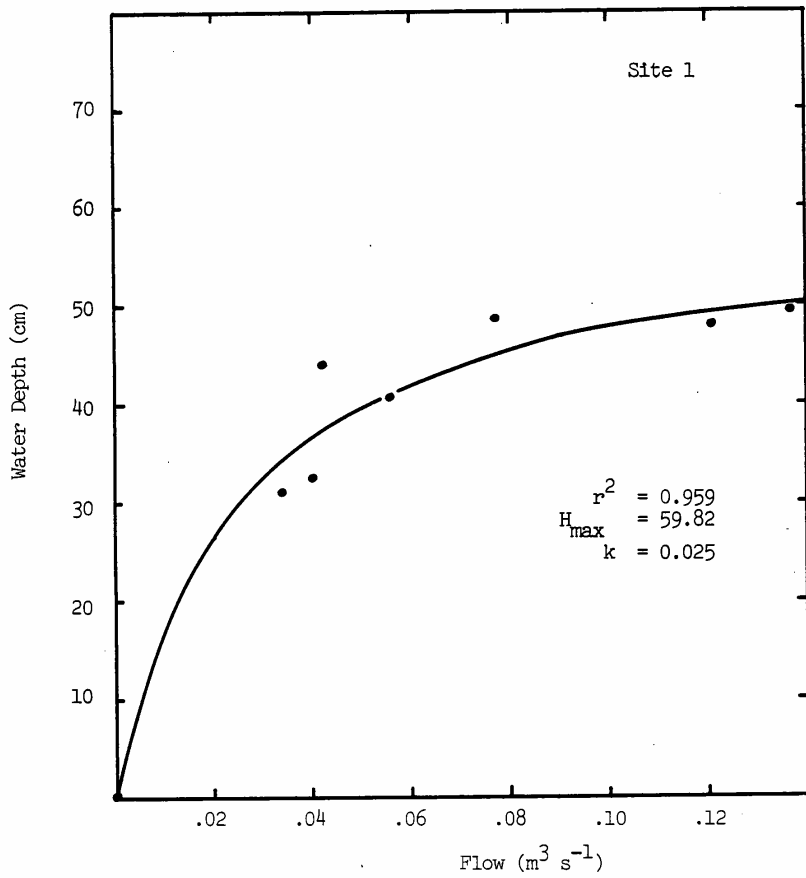


Fig. 2. Discharge curve for tributary  
sampled at Site 2, Lake of the  
Woods.

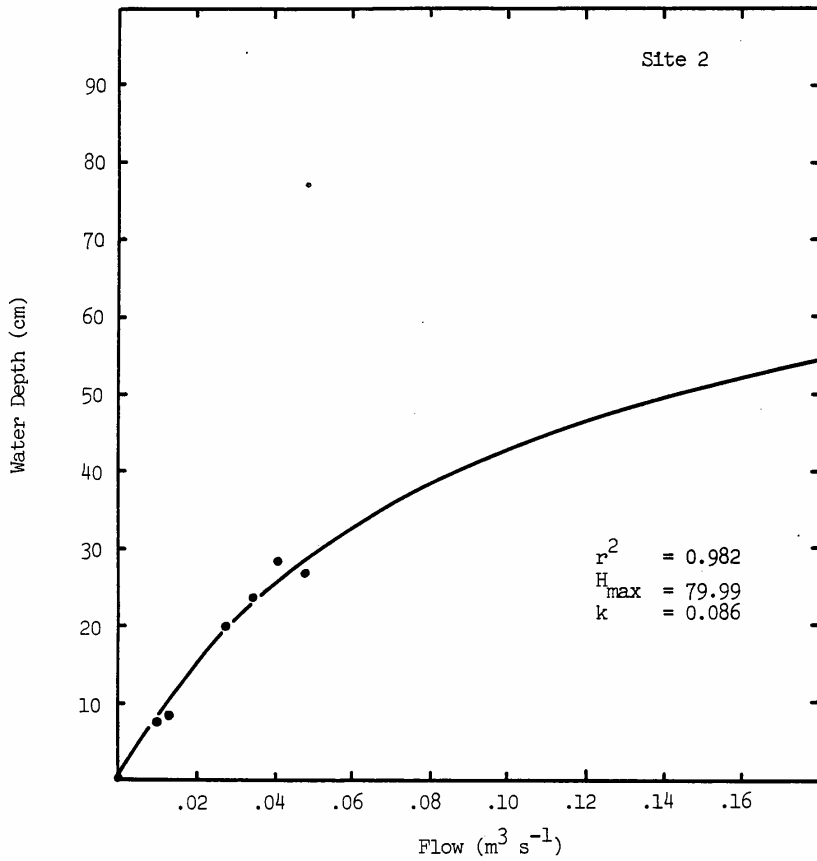


Fig. 3. Discharge curve for tributary  
sample at Site 3, Lake of the  
Woods.

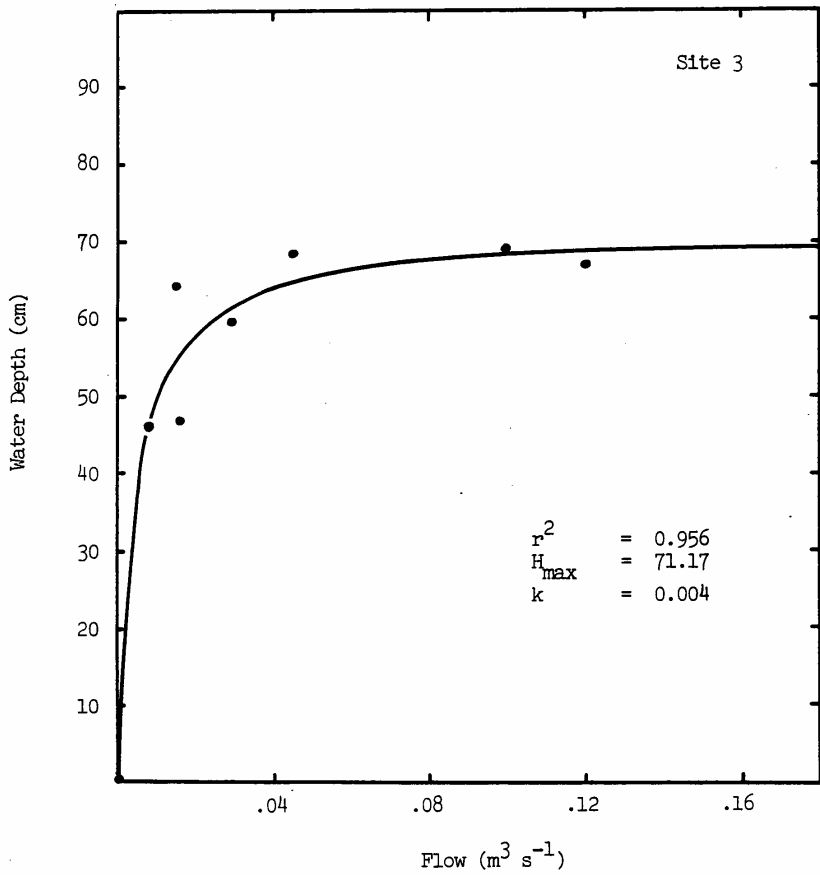


Fig. 4. Discharge curve for tributary  
sampled at Site 4, Lake of the  
Woods.

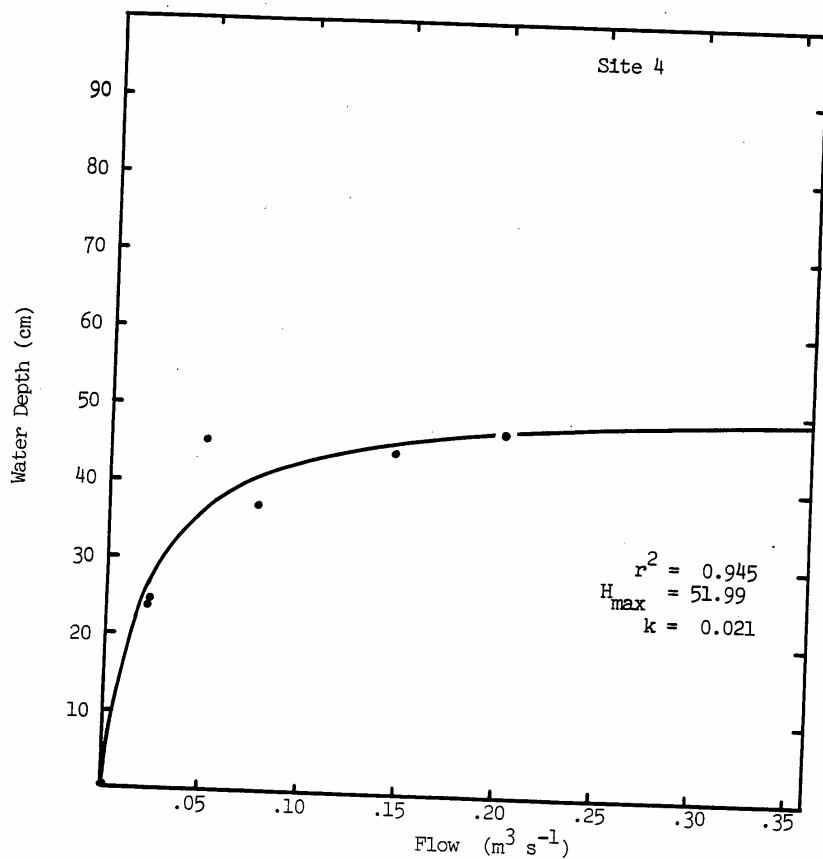


Fig. 5. Discharge curve for tributary  
sampled at Site 5, Lake of the  
Woods.

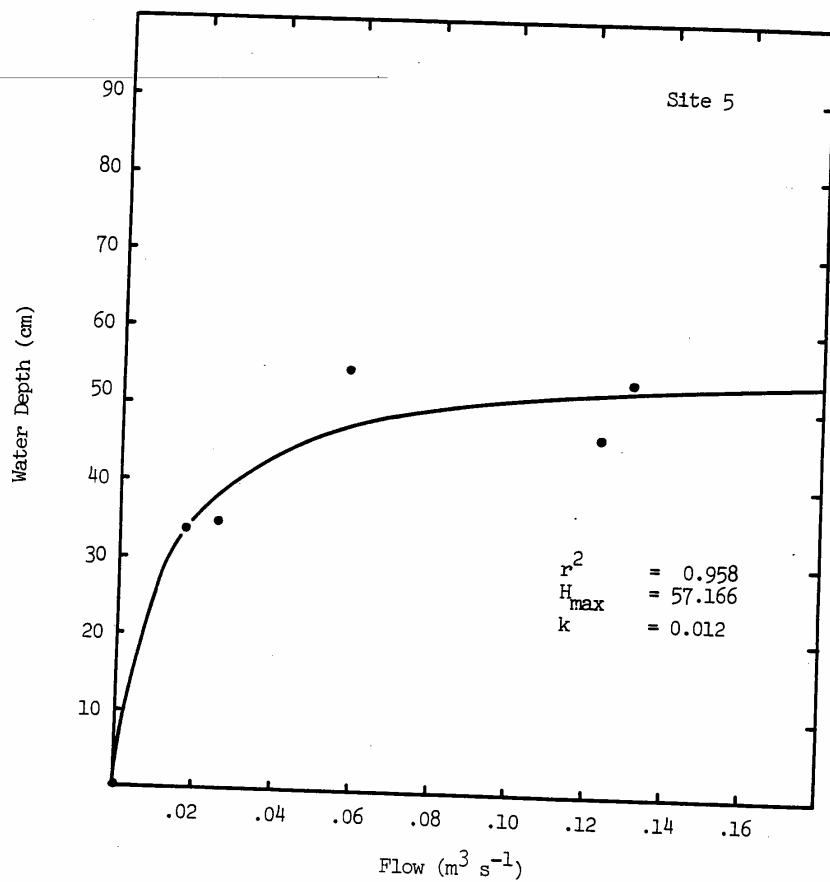


Fig. 6. Discharge curve for tributary  
sampled at Site 6, Lake of the  
Woods.

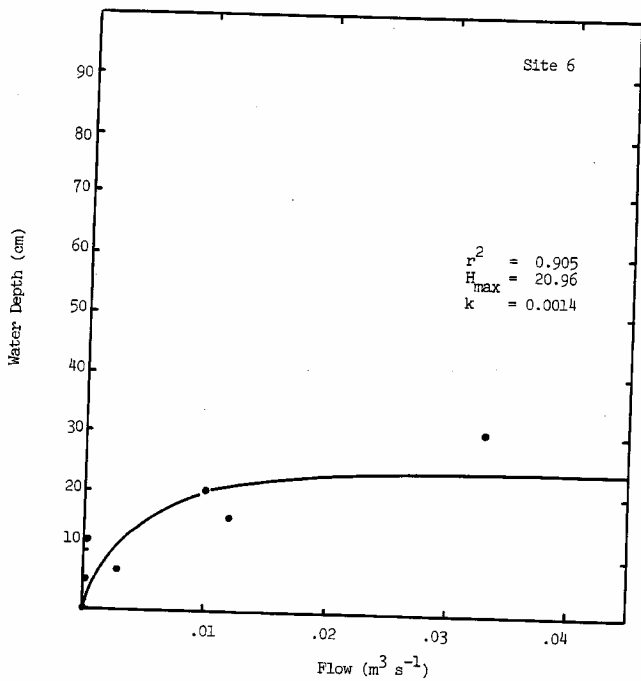


Fig. 7. Discharge curve for the  
Outlet of Lake of the Woods.

Fig. 7

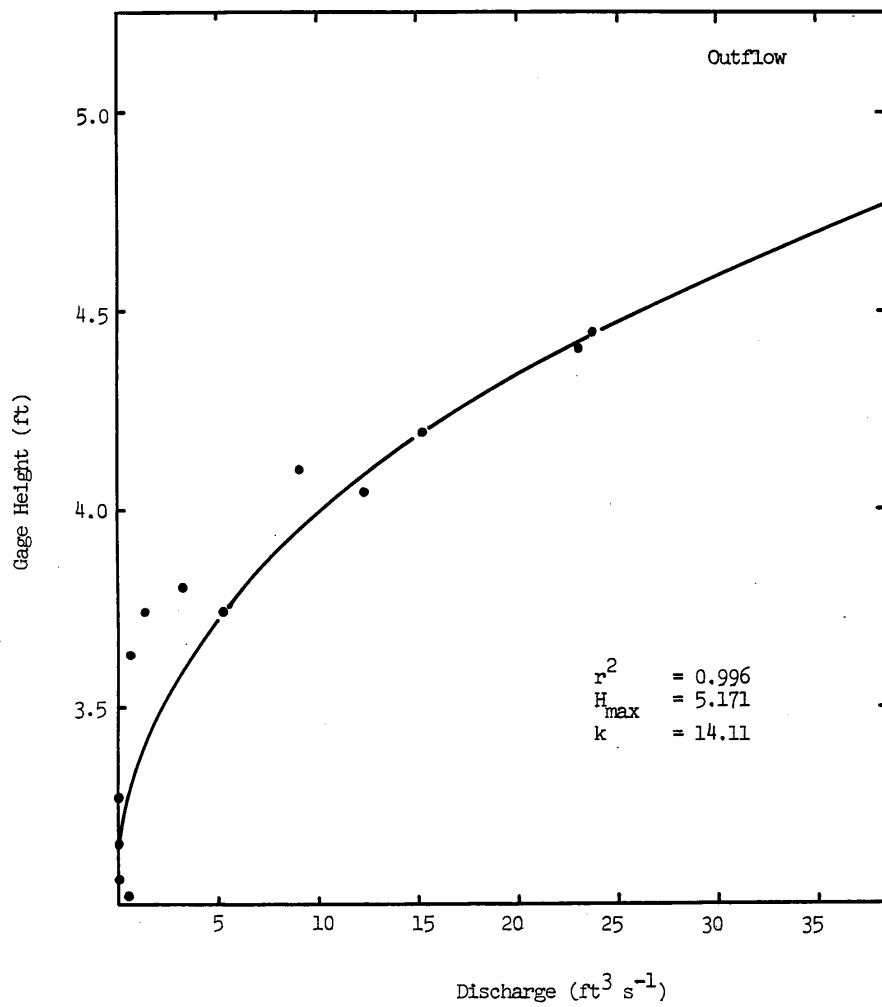


TABLE 1.

Precipitation data for Lake of the Woods.

Month	Total Precipitation (in)
Sept. 1980	5.64
Oct. 1980	3.35
Nov. 1980	1.47
Dec. 1980	3.91
Jan. 1981	0.68
Feb. 1981	1.92
Mar. 1981	0.88
Apr. 1981	5.28
May 1981	6.79
June 1981	6.97
July 1981	3.71
<u>Aug. 1981</u>	<u>2.30</u>
Total	42.90

\* Data from National Weather Service Station,  
South Bend, Indiana.

## APPENDIX V

PREDICTION OF LAKE OF THE WOODS PHOSPHORUS CONCENTRATION USING  
MODELING AND ERROR ANALYSIS BASED UPON LAND USE INFORMATION.

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## APPENDIX V

PREDICTION OF LAKE OF THE WOODS PHOSPHORUS CONCENTRATION USING  
MODELING AND ERROR ANALYSIS BASED UPON LAND USE INFORMATION

## CONSTRUCTION OF MODEL

Reckhow and Simpson (1980) have developed a mathematical model for estimating the annual average phosphorus concentration for lakes. The model is based upon Reckhow's (1979) work with 47 north temperate lakes sampled as part of the EPA's National Eutrophication Survey (NES).

The basic model is:

$$P = \frac{L}{V_s + q_s} \quad (1)$$

where  $P$  = phosphorus concentration ( $\text{mg } \ell^{-1}$ ),  $L$  = areal phosphorus loading ( $\text{g m}^{-2} \text{ y}^{-1}$ ),  $V_s$  = apparent settling velocity of phosphorus ( $\text{m y}^{-1}$ ), and  $q_s$  = areal water loading ( $\text{m y}^{-1}$ ). From analysis of the NES data, it was found that equation (1) could be rewritten as:

$$P = \frac{L}{11.6 + 1.2 q_s} \quad (2)$$

Hence, mean annual phosphorus concentration for Lake of the Woods can be predicted by determining areal phosphorus loading,  $L$ , and areal water loading  $q_s$ . (Note: The limitations on this model, as discussed by Reckhow and Simpson, did not impair its use for Lake of the Woods).

The areal water loading,  $q_s$ , is determined from the following equation:

$$q_s = \frac{Q}{A_o} \quad (3)$$

where  $Q$  is the inflow water volume ( $\text{m}^3 \text{y}^{-1}$ ) and  $A_o$  is the lake surface area ( $\text{m}^2$ ). The inflow water volume,  $Q$ , is determined as follows:

$$Q = (A_d \times r) + (A_o \times \text{Pr}) \quad (4)$$

where  $A_d$  is the watershed area ( $\text{m}^2$ ),  $r$  is the total annual unit runoff ( $\text{m y}^{-1}$ ), and  $\text{Pr}$  is the annual net precipitation ( $\text{m y}^{-1}$ ).

Parameters used in the calculations of  $q_s$  for Lake of the Woods were:

$$A_o = 1.6429 \times 10^6 \text{ m}^2 \quad (\text{direct calculation from planimetry of lake map})$$

$$A_d = 24.5005 \times 10^6 \text{ m}^2 \quad (\text{MACOG 1978})$$

$$r = 0.2921 \text{ m y}^{-1} \quad (\text{Reussow and Rohne, 1975})$$

$$\text{Pr} = 0.9017 \text{ m y}^{-1} \quad (\text{Reussow and Rohne, 1975})$$

$q_s$  becomes:

$$q_s = \frac{24.5005 \times 10^6 \times 0.2921 + (1.6429 \times 10^6 \times 0.9017)}{1.6429 \times 10^6} \quad (5)$$

$$q_s = 5.2578 \text{ m y}^{-1}$$

The areal  $P$  loading to Lake of the Woods was found using the formula:

$$L = \frac{M}{A_o} \quad (6)$$

where  $M$  is the total mass loading of  $P$  ( $\text{kg y}^{-1}$ ). The total mass loading of phosphorus to the lake was found by summing the contributions from each identifiable source in the lake watershed. For our modeling purposes, the following sources were identified: forested land, agricultural land, urban land, precipitation, dry fallout, miscellaneous and septic tanks. No point source input was included in the model. We express mass loading,  $M$ , as:

$$M = (E_f \times A_f) + (E_{ag} \times A_{ag}) + (E_u \times A_u) + (E_p \times A_o) + (E_d \times A_o) + (E_m \times A_m) + (E_{st} \times \text{number of capita-years} \times (1-SR)) \quad (7)$$

where

$E_f$  = export coefficient for forest ( $\text{kg } 10^6 \text{ m}^{-2} \text{ y}^{-1}$ )

$A_f$  = area of forest land ( $\text{m}^2$ )

$E_{ag}$  = export coefficient for agricultural land  
( $\text{kg } 10^6 \text{ m}^{-2} \text{ y}^{-1}$ )

$A_{ag}$  = area of agricultural land ( $\text{m}^2$ )

$E_u$  = export coefficient for urban land ( $\text{kg } 10^6 \text{ m}^{-2} \text{ y}^{-1}$ )

$A_u$  = area of urban land ( $\text{m}^2$ )

$E_p$  = export coefficient for precipitation ( $\text{kg } 10^6 \text{ m}^{-2} \text{ y}^{-1}$ )

$A_o$  = area of lake surface ( $\text{m}^2$ )

$E_d$  = export coefficient for dry fallout ( $\text{kg } 10^6 \text{ m}^{-2} \text{ y}^{-1}$ )

$E_m$  = export coefficient for miscellaneous land ( $\text{kg } 10^6 \text{ m}^{-2} \text{ y}^{-1}$ )

$A_m$  = area of miscellaneous land ( $\text{m}^2$ )

$E_{st}$  = export coefficient for septic tanks impacting lake  
( $\text{kg capita-year}^{-1} \text{ y}^{-1}$ )

No. capita-years = total septic systems impacting lake

SR = soil retention coefficient (dimensionless)

Selection of the appropriate values for the export coefficients in equation (7) was a difficult task. Reckhow and Simpson (1980) have compiled published data into a table consisting of low, high, and middle (most-likely) estimates for the export coefficients. These are summarized below: (all as  $\text{kg P } 10^6 \text{ m}^{-2} \text{ y}^{-1}$ , except septic which is  $\text{kg capita-yr}^{-1}$ )

	Forest	Agriculture	Urban	Precipitation	Misc.*	Septic
high	45	300	500	60	172.5	1.8
middle	20	105	190	35	62.5	0.6
low	2	10	50	15	6.0	0.3

\*found by averaging forest and agricultural values.

Watershed usage areas with their sources in parentheses are:

$$A_r = 2.3521 \times 10^6 \text{ m}^2 \quad (\text{MACOG 1978})$$

$$A_{ag} = 20.5559 \times 10^6 \text{ m}^2 \quad (\text{MACOG 1978})$$

$$A_u = 0 \quad (\text{MACOG 1978})$$

$$A_m = 1.5925 \times 10^6 \text{ m}^2 \quad (\text{MACOG 1978})$$

$$A_o = 1.6429 \times 10^6 \text{ m}^2 \quad (\text{from planimetry of topographic map})$$

The number of capita years was calculated as:

$$\begin{aligned} \text{Number of capita years} = & \text{average \# persons} \times \text{\# days spent at} \quad (8) \\ & \text{per unit} \times \frac{\text{unit per year}}{365} \times \text{\# units} \end{aligned}$$

Data provided by the Lake of the Woods Lake Association yields the following estimate:

$$\text{Total capita - years} = (4 \times \frac{365}{365} \times 375) = 1500$$

Using the low, high, and most-likely values for the export coefficients, the following estimates of mass loading to Lake of the Woods were found:

$$M_{\text{high}} = (45 \times 2.3521) + (300 \times 20.5559) + (500 \times 0) + (60 \times 1.6429) + (172.5 \times 1.5925) + 1.8 \times 1500 \times 1.0$$

$$M_{\text{high}} = 9477.3 \text{ kg P y}^{-1} \quad (9a)$$

$$M_{\text{ml}} = (20 \times 2.3521) + (105 \times 20.5559) + (190 \times 0) + (35 \times 1.6429) + (80 \times 1.6429) + (62.5 \times 1.5925) + (.6 \times 1500 \times 1.0)$$

$$M_{\text{ml}} = 3393.9 \text{ kg P y}^{-1} \quad (9b)$$

$$M_{\text{low}} = (2 \times 2.3521) + (10 \times 20.5559) + (50 \times 0) + (15 \times 1.6429) + (80 \times 1.6429) + (6 \times 1.5925) + (.3 \times 1500 \times 1.0)$$

$$M_{\text{low}} = 825.9 \text{ kg P y}^{-1} \quad (9c)$$

In all cases, a soil retention coefficient of 1.0 was used (this is a worst case scenario which assumes that 100% of P from the septic tanks reaches the lake). Plugging the values obtained from equations (9a-c) into equation (6) yields loading estimates of:

$$L_{\text{high}} = \frac{9477.3}{1.6429 \times 10^6} = 5.77 \text{ g m}^{-2} \text{ y}^{-1} \quad (10a)$$

$$L_{\text{ml}} = \frac{3393.9}{1.6429 \times 10^6} = 2.07 \text{ g m}^{-2} \text{ y}^{-1} \quad (10b)$$

$$L_{\text{low}} = \frac{825.9}{1.6429 \times 10^6} = 0.50 \text{ g m}^{-2} \text{ y}^{-1} \quad (10c)$$

These loading estimates were used along with the value of  $q_s$  from equation (5) to provide a prediction of the in-lake P concentration in Lake of the Woods by applying equation (2):

$$P_{\text{high}} = \frac{5.77}{11.6 + 1.2 (5.258)} = 0.322 \text{ mg } \ell^{-1} \quad (11a)$$

$$P_{\text{ml}} = \frac{2.07}{11.6 + 1.2 (5.258)} = 0.116 \text{ mg } \ell^{-1} \quad (11b)$$

$$P_{\text{low}} = \frac{0.50}{11.6 + 1.2 (5.258)} = 0.028 \text{ mg } \ell^{-1} \quad (11c)$$

The uncertainty of these predictions was estimated by non-parametric first-order error analysis as detailed by Reckhow and Simpson (1980). Accordingly, the 80% confidence limits on the most-likely in-lake P concentration become:

$$0.0365 \text{ mg } \ell^{-1} < P < 0.2818 \text{ mg } \ell^{-1} \quad (12)$$

#### VERIFICATION OF MODEL

The estimated value of  $q_s$  from calculations was  $5.258 \text{ m y}^{-1}$ . The measured value of  $q_s$  for the year beginning September 1980 through August 1981 was  $4.07 \text{ m y}^{-1}$ . Causative factors of this slight overestimation are unclear, but may be related to the extensive drainage network in the area surrounding the lake. Artificial drainage ditches may channel water away from the lake, making the effective watershed area somewhat smaller than the measured watershed area. This could account for a large portion of the discrepancy.

The measured mass phosphorus loading,  $M$ , was  $1012 \text{ kg P y}^{-1}$ . This value falls in between the low ( $825.9 \text{ kg P y}^{-1}$ ) and most-likely ( $3393.9 \text{ kg P y}^{-1}$ ) estimates of mass loading. Since most of the  $P$  entering the lake from external sources originated in the watershed export compartment, an overestimate of the effective watershed area as mentioned above could account for the large value of  $m_{ml}$ . The measured areal external  $P$  loading (equation 10c, adjusted for  $M$ ) was  $0.62 \text{ g m}^{-2} \text{ y}^{-1}$ .

The predicted mean annual in-lake phosphorus concentration ( $P$ ) using the measured values of  $L$  ( $0.62 \text{ g m}^{-2} \text{ y}^{-1}$ ) and  $q_s$  ( $4.07 \text{ m y}^{-1}$ ) was calculated using equation 2 as:

$$P = \frac{0.62}{11.6 + 1.2(4.07)} \quad (13)$$

$$P = 37.4 \text{ } \mu\text{g l}^{-1}$$

The measured mean annual phosphorus concentration averaged over all depths and dates was  $64.9 \text{ } \mu\text{g l}^{-1}$ . Comparison of the two figures reveals a large disagreement, the measured value much higher than the predicted. This is no doubt due to the high internal  $P$  loading from the sediments, a source not considered in the model. This relationship, when used with values corrected for internal loading, is important in predicting the response of Lake of the Woods to various hypothetical reductions in phosphorus loading. Implications are discussed in the management plan section (Section IX).

## APPENDIX VI

## PHOSPHORUS COMPARTMENTALIZATION OF LAKE OF THE WOODS

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## APPENDIX VI

## PHOSPHORUS COMPARTIMENTALIZATION OF LAKE OF THE WOODS

All of the phosphorus retained in lacustrine systems becomes incorporated into one of the following major phosphorus pools:

1. the fish biomass
2. the macrobenthos biomass
3. the macrophyte biomass
4. the seston - includes living and dead particulate suspended matter (i.e. zooplankton, phytoplankton, bacteria, etc.)
5. dissolved phosphorus in the lake water
6. the sediments

The activity of biotic and abiotic factors in the lake ecosystem cause a continual flux of phosphorus from one pool to another. The differential response of various biotic populations to changing ambient environmental conditions induces short-term variation in the size of these pools. An overall homeostatic condition subject to temporal fluctuations prevails, and unless interrupted, maintains a relatively constant amount of phosphorus in each of the biotic compartments on a year to year basis. Very gradual increases do occur, however, as the lake undergoes the process of eutrophication. Typically, most of the phosphorus entering the lake system eventually reaches the sediment pool.

Estimates of the relative magnitude of the various phosphorus compartments of Lake of the Woods are given in Table 1. The derivation of these values are discussed in the following material.

The fish standing stock in Lake of the Woods was estimated to be  $336.24 \text{ kg ha}^{-1}$  (Stu Shipman, I.D.N.R., personal communication). The live weight phosphorus content of fish is 1.99% (Applegate 1971). The fish community, therefore, contains  $6.69 \text{ kg P ha}^{-1}$ . This value times the surface area of the lake yields a total phosphorus pool of  $1099 \text{ kg P}$  in the fish community.

Macrobenthos biomass estimates were made using the data supplied by Wohlschlag (1950). Based on his data, the average benthic biomass was calculated to be  $764.4 \text{ mg dry weight m}^{-2}$ . Insects have an average dry weight phosphorus content of 1.0% (Spector 1956). This data allows calculation of an areal macrobenthic phosphorus content of  $7.64 \text{ mg P m}^{-2}$ . The total amount of phosphorus in the macrobenthos pool is therefore found by multiplying the areal phosphorus content by the surface area of the sediments ( $1.9715 \times 10^6 \text{ m}^2$ ) to yield  $15 \text{ kg P}$ .

Macrophyte biomass and tissue nutrient content were monitored during the course of the study (see Figs. 24 and 25 of Appendix III). The largest standing stock of aquatic vascular plants occurred in early June, 1981. The amount of phosphorus in the plant tissue nutrient pool at this time was calculated to be about  $40 \text{ kg P}$ . This value was selected for all subsequent analysis, and represents the maximum expected phosphorus content in this compartment.

The particulate phosphorus in the seston and the dissolved phosphorus in the lakewater were collectively quantified in the total phosphorus analysis of in-lake water samples. The weighted average phosphorus concentration, calculated over all depths and sampling dates, was  $64.9 \text{ } \mu\text{g l}^{-1}$ . This value was multiplied by the lake volume ( $7.848 \times 10^9 \text{ l}$ ) to estimate the magnitude of this phosphorus

pool. Results of this analysis indicate that 509 kg P are contained in these combined phosphorus compartments.

The sediment phosphorus pool was directly quantified as part of the overall investigation (see section VI). This compartment contained 16,393 kg P, and overwhelmingly represented the greatest store of phosphorus in the entire system.

TABLE 1.  
Phosphorus compartments of  
Lake of the Woods.

Compartment	kg P	% of Total
Fish	1099	6.1
Macrobenthos	15	0.1
Macrophytes	40	0.2
Seston/dissolved P	509	2.8
Sediments	16,393	90.8